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The first meeting was held in Corvallis, Oregon in November 1975 and the second was hosted by the Japanese Government in October 1976. The third session was convened in November 1977 in Easton, Maryland and the fourth session in Tokyo. The fifth meeting (at which these papers were presented) was held in New Orleans, Louisiana.

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The United States-Japan Ministerial Agreement of May 1974 provided for the exchange of environmental information in several areas of mutual concern. This report is the compilation of papers presented at the Fifth United States-Japan Experts' Meeting on the Management of Bottom Sediments Containing Toxic Substances, one of the 10 identified problem areas.

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DREDGING ON A COMPETITIVE BASIS

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ABSTRACT

This paper describes the dredging mission of the U.S. Army Corps of Engineers to construct and maintain adequate dimensions in navigation projects to accommodate maritime traffic and the scope of the national dredging program required to accomplish this mission. A decline in the workload or dredging yardage levels during the period of 1963 through 1978 and the adverse effects of this decline are discussed. The paper outlines the factors relating to the evolution of Public Law 95-269 and the major provisions of this legislation which was enacted April 26, 1978. The Industry Capability Program, which was initiated by the U.S. Army Corps of Engineers in December 1976, is described. This program, which provides an opportunity for industry dredges to compete with Corps dredges, was implemented to encourage the industry to make the large capital outlays required to construct new dredges, particularly seagoing hopper dredges. The statistics accumulated from the Industry Capability Progam (ICP) are presented and summarized. In addition, the industry construction program which has evolved due to the ICP is discussed. Also, the paper describes the "minimum fleet" of the U.S. Army Corps of Engineers as provided for in Public Law 95-269 and the factors considered in developing a recommendation to the Congress as to the number and class of hopper dredges which would comprise the "minimum fleet."

INTRODUCTION

Public Law 95-269 directs the Chief of Engineers, U.S. Army, to undertake a study to be submitted to the Congress for the purpose of determining the minimum Federally-owned fleet of dredges required to perform emergency and national defense work. This law also provides that as the industry demonstrates its capability to perform the dredging work currently performed by the existing Federally-owned fleet, at reasonable prices and in a timely manner, the existing Federally-owned fleet of dredges shall be reduced by the orderly retirement of plant until the minimum fleet prescribed by the Congress is reached. The legislation also indicates that the minimum fleet of the Corps of Engineers shall be maintained to technologically modern and efficient standards and be kept in a fully operational status. This paper presents the background factors relative to the evolution of Public Law 95-269 and a status report on the Corps and industry efforts to implement the provisions of the legislation. The paper is relative to only the seagoing hopper dredge program of the Corps of Engineers because this is a program in which the industry was not active until 1977. In addition, a study of this program has been completed and the Chief of Engineers recommendation as to the number and class of hopper dredges which would comprise the Federally-owned minimum fleet has been presented to the Secretary of the Army and the Office of Management and Budget. A study of the non-hopper dredge elements of the minimum fleet is nearing completion and is scheduled for presentation to the Chief of Engineers in the next month or so.

NAVIGATION MISSION

In 1824 Congress assigned the U.S. Army Corps of Engineers the responsibility for improving and maintaining the navigation channels of the nation's ports, harbors and inland waterways. Since that time, the Corps of Engineers has taken part in the construction, maintenance and improvement of over 25,000 miles of navigable waterways.

These waterways serve 130 of the nation's 150 largest cities and are utilized to transport one-fourth of the nation's ton-miles of domestic cargo. Thus, they are essential to the economic well-being of the nation. Nearly 60 percent of our waterways are also vital to our ability to meet the energy needs of the country.

There are 107 commercial ports and 416 small boat harbors that include Federally-authorized channels. The ports and harbors of the nation handle nearly two billion tons of cargo annually and serve over seven million recreation craft.

The maintenance and improvement of the waterways to make them suitable for waterborne commerce is one of the major responsibilities of the Civil Works program of the Corps of Engineers. During the past three years an average of 308 million cubic yards were dredged at an average annual cost of about $228 million. The major part of the annual dredging work (about 95%) is accomplished using cutterhead, dustpan and seagoing hopper dredges. The remaining 5% of the annual dredging workload is accomplished by the use of bucket, dipper and sidecasting dredges. In the case of the lower and mid sections of the Mississippi River and tributaries the work is performed primarily with the hydraulic dustpan type of equipment since these dredges were designed especially to operate under conditions unique to these waterways.

The Corps of Engineers accomplishes the majority of the annual workload by utilizing industry equipment under competitive bidding procedures and performs the remaining work (about 35%) with Corps-owned dredges. Presently, the Corps operates a fleet of 36 dredges which remove about 128 million cubic yards of material annually at a cost of $86 million. The industry owns about 476 dredges and performs about 180 million cubic yards of dredging annually at a cost of $151 million.

LEGISLATIVE BACKGROUND

Prior to the enactment of Public Law 95-269 the dredging program of the Corps of Engineers was accomplished under the provisions of 33 USC 622 and 33 USC 624. These laws required that the dredging workload be performed in the most economical or advantageous manner by use of either Corps dredging plant or by industry plant. Public Law 95-269, which replaced the above cited statutes, includes similar language indicating that the Corps of Engineers will utilize contractor equipment when industry reasonably demonstrates its capability to perform the work done by the existing Federally-owned fleet at reasonable prices and in a timely manner. This legislation also includes the following provisions:

-- That a study be undertaken by the Corps of Engineers to determine the minimum Federally-owned fleet required to perform emergency and national defense work. The legislation indicates that the study is to be submitted to the Congress within two years after the April 26, 1978 enactment of Public Law 95-269.

-- That no river and harbor improvement work shall be done by private contract if Federally-owned plant is reasonably available to perform the work and the contract price is more than 25 per centum in excess of the estimated comparable cost of doing the work with Corps plant.

-- That when Corps plant is not reasonably available no river and harbor improvement work shall be done by private contract if the contract price is more than 25 per centum in excess of a fair and reasonable cost of a well-equipped contractor doing the work.

-- That the Corps of Engineers may retain as much of the existing Federally-owned fleet as long as necessary to insure the capability of the Corps of Engineers and industry to carry out projects for improvement of rivers and harbors.

-- That the Corps of Engineers shall retain a technologically modern minimum fleet of dredges to carry out emergency and national defense work and that this fleet shall be kept in a fully operational status.

FACTORS LEADING TO ENACTMENT OF PUBLIC LAW 95-269

Reduced Scope of Capital Improvement Projects Since World War II

Since World War II there have been few instances in which the channels of the ports and inland waterways of the country have been widened and deepened to any significant extent. While there have been many deep draft channels and harbors constructed for large supertankers and bulk cargo ships in many parts of the world, there have been no such facilities constructed in the United States since World War II. Thus, the dredging industry of the United States has not had an opportunity to engage in large and lucrative dredging operations such as those at Rotterdam, The Netherlands; Zeebrugge, Belgium; Dunkirk, Le Havre and Gulf de Fos, France; and Botany Bay, Australia. Due to the decline in the scope of the annual workload since World War II, the financial condition of many of the United States dredging firms has deteriorated and some of the large firms have gone out of business.

Annual Improvement Dredging Workload and Expenditures

For the extended period of 1963 through 1978 industry dredges have performed, on an average, 86% of all the improvement or new work workload dredging and by 1978 the industry performed 97% of the new work dredging. The annual improvement dredging decreased significantly from 263 million cubic yards in 1963 to only 70 million cubic yards in 1978. This dramatic reduction in the improvement dredging workload constituted the bulk of the overall decrease in the total dredging workload for this period. Annual expenditures for improvement dredging for this period also decreased from $107 million in 1963 to $93 million in 1978. The unit cost, which is probably the best factor to consider in evaluating cost trends over extended periods, was $0.41/cubic yard in 1963 and $1.33/cubic yard in 1978. The cost/cubic yard for 1978 compares favorably with the 1963 unit cost because it reflects a reasonable average annual escalation of 8%.

Annual Maintenance Dredging Workload and Expenditures

During the 1963 to 1978 period the annual maintenance dredging workload experienced an upward trend with several significant peaks occurring. However, the net result was a slight decrease of 3% when comparing the 1963 maintenance dredging workload of 217 million cubic yards to 210 million cubic yards in 1978. It should be noted, however, that the industry workload during this period averaged 48% of the total maintenance workload. Moreover, the industry workload actually increased from 80 million cubic yards in 1963 to 118 million in 1978. Maintenance dredging expenditures increased substantially to $210 million in 1978 from $59 million in 1963. However, if these expenditures are calculated in terms of constant 1963 dollars and annual escalation of 8%, the unit rate is found to have remained relatively stable.

Total Annual Dredging Workload and Expenditures

The total annual workload or cubic yardage decreased dramatically from 480 million cubic yards in 1963 to 280 million cubic yards in 1978. Most of this decrease occurred from 1963 to 1967 and with the exception of some periodic peaks there was a continuing downward trend to the current level.

Expenditures dipped from $160 million in 1963 to $92 million in 1967 and then climbed to a 1978 total cost of $280 million. Once again, on the basis of unit cost, this increase merely reflects an annual escalation of about 8% over the 15-year period.

Corps/Industry Distribution of the Annual Dredging Workload and Expenditures

The Corps of Engineers has performed the majority of the total annual dredging workload with industry equipment for many years. The Corps/industry percentage distribution of the total annual workload yardage during the period of 1963 to 1978 was 41/59% even though the industry did not have any dustpans during this period and entered the hopper dredge field in 1977. During this period, the Corps/industry percentage distribution of the total annual expenditures was 34/66%. On an outlay basis, the percentage distribution of the expenditures for 1963-1978 has been within the 25-35% Corps and 65-75% industry range cited in a 1974 management/consulting firm report as the most economical and optimum allocation of the dredging program between the Corps and the industry.

Summary--Dredging Workload and Expenditures

In the perspective of a rapid and significant decrease in workload and the relative constancy of unit cost from 1963 through 1978 it is not surprising that the industry in the face of such a financial dilemma was reluctant to invest in new equipment or major improvements to existing equipment without some encouragement from the Corps of Engineers and the Congress, even though the industry enjoyed a substantial and increasing share of the total declining workload.

INDUSTRY CAPABILITY PROGRAM

Background

Based upon information contained in a comprehensive study of the national dredging program completed by a management/consulting firm in 1974, the Chief of Engineers concluded there was a need for a comprehensive program to determine, in a structured manner, the capability of the industry to accomplish dredging work at reasonable prices and in a timely manner. A program to meet this objective was initiated on December 13, 1976 with the issuance of Corps of Engineers Circular EC 1125-2-358. This program, known first as the Testing of the Market program and currently known as the Industry Capability Program, was initiated to accumulate detailed operational and cost information to reflect the relative efficiency of existing Corps dredges as related to the performance of industry dredges.

Opportunity for Industry to Compete with Corps Dredges

The Industry Capability Program provides an opportunity for the industry to bid competitively with all types of Corps of Engineers dredges over a broad spectrum of dredging work. Included in this competitive bidding program are the types of projects traditionally accomplished with specialized Corps plants such as dustpan, hopper and sidecasting dredges. The comprehensive statistical data developed by this program will be used by the Congress to determine the relative relationship of the Corps/industry roles in carrying out the Federal dredging requirements.

Program Results

By the end of August 1979 the "Industry Capability Program" yielded the following statistics:

-- 84 projects, with an estimated value of $86.0 million, have been advertised to the industry since the beginning of the program in December 1976.

-- 42 projects have been awarded to the industry at a total contract cost of $52.6 million. This figure was $733,000 below the total cost estimated by the Corps of Engineers to perform the work with Corps dredges.

-- 41 projects have been awarded to the Corps dredges at a total cost of $32.9 million. The industry did not offer any bids on 12 of these projects. The Corps estimate for the 29 projects on which the industry competed was $20.5 million. This was $15.4 million less than the industry bids to accomplish the work.

-- 1 project, with an estimated cost of $483,443, is being protested by the industry. Therefore, the award of this work has been delayed.

-- Discounting the 12 projects which were awarded to the Corps when the industry did not offer any bids, there has been a saving of about $16.1 million to the taxpayers.

To date, the industry equipment has performed reasonably well. Generally, the work has been done satisfactorily, although there have been some isolated problems with the industry hopper dredges and the industry pipeline dredges while working in exposed and ocean waters. After the dredging industry modernizes its fleet and has more experience with hopper dredges, we expect that the industry performance will improve.

A preliminary evaluation of the results of the Industry Capability Program indicates that the industry has risen to the opportunity provided by this program with five hopper type vessels in operation and four hopper dredges under construction. In addition, the industry has constructed one dustpan dredge. Thus, the industry will soon have an increased capability and will be able to compete in a wider range of projects in the future.

Extension of the Program

The program was initially scheduled for completion in December 1980. However, it has been extended to September 1981 to be consistent with the language contained in the Conference Report on the fiscal year 1979 Appropriations Act. This report indicates that a reassessment of the dredging program is to be made by the Corps of Engineers in fiscal year 1982 to determine whether industry's response warrants an increase or decrease in the 30 million cubic yard industry annual hopper dredge workload target contained in the Conference Report.

CONSTRUCTION OF HOPPER DREDGES BY THE INDUSTRY

Background

As a result of the encouragement provided by the Industry Capability Program initiated by the Corps of Engineers and Public Law 95-269, enacted by the Congress on April 26, 1978, the industry has embarked on a program to construct seagoing hopper dredges.

Existing Industry Hopper Dredges

The existing industry hopper fleet of five dredges, as of October 1979, was as follows:

Large Class

One large class hopper dredge, the Long Island, is available. This vessel was built by the Construction Aggregates Corporation in 1971 and acquired by the Great Lakes Dredge and Dock Company in 1978. The Long Island is a barge with a volumetric hopper capacity of 16,000 cubic yards. It is propelled by a tug fitted into a notch in the stern of the barge and is equipped with dual pumps and dragarms. This vessel was initially equipped for only direct pumpout operations. However, it was modified in fiscal year 1978 to include a bottom gate dumping capability which will improve its versatility. The Coast Guard is currently assessing the condition of all types of dredges to determine whether they meet the design standards of the Seagoing Barge Act (USC 395). It is possible that the Coast Guard review will result in a decision that the current load line assigned to the Long Island must be revised. If this is the case, it could reduce the load carrying capability of the vessel from 16,000 cubic yards to a much lesser figure while the vessel is operating in the offshore zone.

Medium Class

Three medium class hopper dredges are available as follows:

Manhattan Island: This vessel, which has dual dredge pumps and dragarms, is owned by the North American Trailing Company (a consortium consisting of the Great Lakes Dredge and Dock Company and Ballast-Needham, a Dutch firm). The dredge was commissioned in June 1977. It is a new and modern hopper dredge with a split hull. It has a volumetric hopper capacity of 3,600 cubic yards and has performed well on the navigation projects on which it has worked for the Corps of Engineers. This vessel is not equipped for direct pumpout operations. However, we are informed by representatives of the firm that it will probably be converted to include this capability.

Esperance: This vessel, acquired from a Dutch firm by the Roger J. Au Company, Mansfield, Ohio, is a converted LST which has only one pump and one dragarm. It was placed in service in May 1978 and has worked on two navigational projects in the Great Lakes area. The vessel has a volumetric hopper capacity of 3,600 cubic yards and is equipped for only direct pumpout operations. The production of this dredge was poor and it is currently for sale. It appears doubtful that the dredge will be reactivated for work in the United States in view of its low production record and the potential conflict of its operation with the provisions of the Coastwise Shipping Act.

Sugar Island: This vessel, which has a volumetric hopper capacity of 3,600 cubic yards, is a sister ship to the Manhattan Island and is equipped for direct pumpout operations. The vessel was placed in service in May 1979 and is expected to perform well.

Small Class

Only one small class hopper dredge, the Manson, is available. This vessel, owned by the Manson/Osberg Company, Seattle, Washington, is a barge with a volumetric hopper capacity of 1,600 cubic yards. It is propelled by a tug coupled to the stern of the barge with articulating hydraulic arms and is equipped with only one pump and one dragarm. This vessel was placed in service in July 1978 so the experience base for this vessel is very limited. Thus far it has worked only inside estuaries and has performed well. However, there is some doubt as to its ability to perform well on the entrance or bar channels exposed to the wave action of the Pacific Ocean.

Industry Hopper Dredges Under Construction

The four hopper dredges under construction by the industry are as follows:

Large Class

One large class hopper dredge (about 8,800 cubic yards) is under construction at the Avondale Shipyard in New Orleans, Louisiana. Details on the construction schedule and delivery date are not available at this time. It is understood that the owners of the vessel will be the National Dredging Company (a consortium consisting of Zapata Marine of Houston, Texas, and Bos Kalis, a Dutch firm).

Medium Class

Two medium class hopper dredges are under construction by the industry as follows:

Eagle I: A contract was awarded for the construction of this vessel during September 1978 by the Eagle Dredging Company (a consortium consisting of the C. F. Bean Company of New Orleans and Volker-Stevin, a Dutch firm). We are informed by the owners that this vessel is scheduled for delivery in October 1980. It will have a volumetric hopper capacity of about 4,750 cubic yards, dual pumps and dragarms and a split hull. It is expected to be an efficient dredge representative of current design technology.

Dodge Island: A contract was awarded to Southern Shipbuilding, Slidell, Louisiana for the construction of this vessel during March 1979 by the Great Lakes Dredge and Dock Company, Chicago, Illinois. We are informed by the owners of this vessel, which will be a sister ship to the Manhattan Island and the Sugar Island, that this vessel is scheduled for delivery in the summer of 1980. It will have a volumetric hopper capacity of 3,600 cubic yards, dual pumps and dragarms, a split hull, and is expected to be an efficient dredge representative of current design technology.

Small Class

One small class hopper dredge (about 1,300 cubic yards) is under construction by the Twin City Barge Corporation in St. Paul, Minnesota. It is understood that the owners of the vessel will be a consortium consisting of the T. L. James Company of New Orleans and HAM Holland, a Dutch firm. Delivery of this vessel, which has not been given a name, is scheduled for May/June 1980. We are not aware of the physical and equipment features of this dredge.

Hopper Dredge Construction Planned by the Industry

Large Class

We are not aware of any industry plans to construct additional large class hopper dredges. Our assessment of the national dredging program indicates that there are a limited number of navigation projects which will require large class hopper dredges. Therefore, it seems doubtful that many of these vessels will be constructed.

Medium Class

The T. L. James/HAM Holland consortium has announced plans to award a contract in the near future for the construction of a medium class hopper dredge. It is understood this vessel will have a volumetric hopper capacity of about 3,500 cubic yards. We are not aware of the physical and equipment features of this dredge.

Small Class

The C. F. Bean/Volker-Stevin consortium has announced plans to award a contract in the near future for the construction of a small class hopper dredge, the Eagle II. We are not aware of the volumetric hopper capacity or the physical and equipment features of this dredge.

Summary

Existing Industry Fleet of Hopper Dredges: Consists of five vessels, one large class, three medium class and one small class.

Hopper Dredges Under Construction: There are four hopper dredges under construction by the industry including one large class, two medium class, and one small class.

Hopper Dredge Construction Planned: The construction plans of the industry, as of October 1979 included two vessels, one medium class and one small class.

Total Industry Fleet of Hopper Dredges: As of October 1979, includes eleven vessels, two large class, six medium class and three small class hopper dredges.

CORPS OF ENGINEERS HOPPER DREDGE FLEET

Background

During the period of 1871 to 1906 the industry and the Corps of Engineers operated the first hopper dredges in the world. During 1899 to 1906 the industry experienced a series of problems in the deepening of the Ambrose Channel leading to New York Harbor. Based on these problems, the Congress directed the Corps of Engineers to assign two Corps hopper dredges, the Atlantic and Manhattan in 1902 to the project to expedite the deepening of the channel. Later in 1902 the Congress authorized the Corps of Engineers to design, construct and operate twelve additional hopper dredges. In 1906, the only remaining industry firm operating hopper dredges, Metropolitan Dredging Company, went out of business. From 1906 until 1977 when the industry reacted to the Industry Capability Program, the only hopper dredges available to work on navigation projects in the United States were those operated by the Corps of Engineers.

Existing Fleet

Number and Type

The active Corps of Engineers fleet consists of 14 hopper dredges. Two of these dredges are in the large class (6,000 cubic yards or greater hopper capacity), seven are in the medium class (2,000 to 6,000 cubic yards), and five are in the small class (under 2,000 cubic yards).

Age and Condition

The 14 existing Corps of Engineers hopper dredges have an average age of 32.1 years. The existing fleet is generally obsolete with three of the dredges having an average age of 41 years as compared with a reasonable economic life cycle of 20 to 25 years. Older dredges require extensive maintenance and repair work to keep them operational. The repair of old equipment is expensive and it also results in considerable lost time.

Construction of New Hopper Dredges

The Congress has authorized the construction of the following three new hopper dredges by the Corps of Engineers.

West Coast Shallow Draft Hopper Dredge

This vessel, with a volumetric hopper capacity of 825 cubic yards, is under construction by the Norfolk Shipbuilding and Drydock Company, Norfolk, Virginia. Completion is scheduled for June 1980.

Large Class Hopper Dredge

This vessel, with a volumetric hopper capacity of 8,400 cubic yards, is under construction by the Avondale Shipyard, New Orleans, Louisiana. Completion of construction is scheduled for March 1981.

Medium Class Hopper Dredge

This vessel, with a volumetric hopper capacity of 6,000 cubic yards, is under construction by the Sun Shipbuilding and Drydock Corporation, Chester, Pennsylvania. Completion of construction is scheduled for January 1982.

Summary

These three new dredges will constitute the nucleus of the minimum Federally-owned hopper dredge fleet required to meet the emergency and national defense requirements as provided for in Public Law 95-269.

MINIMUM HOPPER DREDGE FLEET OF THE CORPS OF ENGINEERS

Background

In developing the methodology for determining the hopper dredge requirements of the minimum fleet, paramount consideration was given to the national defense mission cited in Public Law 95-269. A study of the national defense needs was prepared by the Engineer Studies Center (ESC). This study focused on determining the number the dredges needed to support military operations in the continental United States and overseas. The study findings were predicated on an evaluation of the Department of Defense planning guidance, current war and contingency operational plans and a wide range of operational and logistical contingencies. The ESC study concluded that seven hopper dredges were necessary to support the military operations of the nation, including three medium and four small class vessels. After reviewing the hopper dredge fleet required to provide for the military mission, the emergency dredging needs of the nation were evaluated by the staff of the Civil Works Directorate. This evaluation indicated that a fleet of eight hopper dredges was needed and that a change in the mix of the types of dredges was also required. The assessment of the Civil Works staff indicated that eight hopper dredges, consisting of one large class, four medium class and three small class vessels, could meet both the military and emergency dredging needs of the nation. However, this assessment also indicated that this fleet of eight hopper dredges could not meet both of these needs on a simultaneous basis.

Factors Considered in Determining the Minimum Hopper Dredge Fleet

The major factors considered in the review of the hopper dredge requirements of the minimum fleet were as follows:

-- Geographical distribution of the navigation projects in the United States and the overseas deployment areas indicated in the Engineer Studies Center report.

-- Project dimensions and operational conditions as related to the sizes and types of hopper dredges needed. Large and medium class hopper dredges cannot gain access to shallow draft navigation projects. Conversely, small class hopper dredges are either ineffective or very inefficient while operating in deep draft channels.

-- The frequency of the dredging cycle required at each project, i.e, biannual, annual or multi-year cycles. The frequency of the dredging cycle is a major factor in evaluating the capability of immediate response.

-- The level of maintenance required on each of the projects. In those cases where the level of maintenance is minimal, i.e., when the depth provided is only marginally greater than the depth required for marine traffic, it is often necessary to dredge the channel more than once each year.

-- Projects which have a rapid and extreme shoaling rate. In the lower Mississippi River and other major delta regions, it is not unusual for the shoaling rate during the runoff season to reduce the flotation depths by several feet in a matter of a few days. An immediate response capability is required in these situations to restore normal navigation.

-- The haul distances from the navigation projects to the disposal areas. During the past ten years there has been an increase in the distances from the dredging areas to the disposal areas. This leads to an increase in the dredge production time required per project and relates to the number and size of hopper dredges required. This factor also leads to an increase in the annual funding requirement and in the unit cost per cubic yard.

-- The dredging depths at the various projects. This factor must be considered in determining the number and size of dredges required. The small class dredges are not equipped with dragarms long enough to dredge the deep draft projects. Large class and medium class dredges cannot function efficiently in shallow draft areas.

-- The type of material at the various projects. A variety of materials must be dredged ranging from light silts to heavy sands and gravel. The type of material affects the productive capability of hopper dredges. For example, the excavation of gravel is difficult and time consuming which results in an increase in the dredge production time required at the projects. On the other hand, light materials can also extend the requirement for dredge production time when the materials do not settle in the bins of the hopper dredges and reduce the efficiency of the operations.

-- The requirement for direct pumpout operations. In certain situations it is necessary that the material excavated by hopper dredges be unloaded in diked disposal areas. This type of operation is known as the direct pumpout dredging mode. This operation results in a large increase in the dredge production time required due to two factors. First, the hopper dredge must travel to a designated location and be coupled to a mooring facility. Then the material must be pumped through a discharge line into the diked area. Based on the type of mooring facility, the exposure to wave action, the distance from the dredging area to the mooring facility and the length of the discharge pipeline, the total cycle time required ranges from two to four times greater than the cycle time required for open dumping operations. There has been a great increase in the use of the direct pumpout mode in the past twenty years due to environmental considerations and beach nourishment requirements. It is expected that there will be a further increase in this trend.

-- Limitations in the periods when dredging operations can be conducted. There are two factors which lead to this situation. First, there are areas in which dredging operations can be conducted only during given months due to environmental considerations such as the spawning seasons for marine life species. Secondly, there are areas in which the wave conditions are so severe that dredging operations cannot be conducted during certain months of the year. These conditions, which lead to a concentration of dredge production time during a limited period, have a bearing on the number and size of hopper dredges required to meet the emergency dredging requirements.

-- Restrictions on overflow dredging. In several areas there are environmental considerations, such as spawning seasons and the presence of polluted materials, which eliminate the use of the overflow method of dredging. In many cases, the excavated materials are pumped into the bins of the hopper dredges past the capacity of the hoppers of the dredge. This technique increases the volume of material that can be carried during each dredging cycle and improves the efficiency of the operations. Restrictions on the use of this dredging mode result in an increase in the production time required and has a bearing on the number and size of dredges required to meet emergency conditions.

-- The operating schedule of hopper dredges. Corps of Engineers hopper dredges are operated on a twenty-four hours per day, seven days per week schedule for eleven months per year. The twelfth month is reserved for major repairs and overhauls. By staggering the repair schedules the majority of the fleet of dredges are in operation at any given time. It is planned that this type of schedule will be followed when the minimum fleet is available in order to provide for the optimum use of the available hopper dredges.

-- The transit time required to move from one location to another. The distances in each of the three coastal regions and the Great Lakes are in the range of 1,000 to 1,500 miles. Therefore, the frequency of the dredging cycles and the proximity of one emergency situation to another can result in an extended transit period. This factor must be considered in determining the number and size of dredges required to meet emergency conditions. If the regional assignment of dredges were not provided the transit times between various regions would be in the range of three to four weeks.

-- The effective time rate of the hopper dredges. Effective time is that spent during the actual dredging operations, including the pumping, loading, hauling and disposal cycles. The time spent transiting between project locations, taking on fuel and supplies, repairs, delays due to weather and all other non-productive operations is non-effective time. Therefore, the lesser the number of dredges available, the greater the percentage of time that must be spent in traveling between project locations and the greater the nonproductive time.

-- The collision and sinking of hopper dredges. The Corps of Engineers records indicate that a hopper dredge is lost through sinking once every ten years. In addition, on the average, the number of collisions with other ships and groundings is in the range of two to three per year. In most cases the damages sustained are not major. However, lost time for repairs occurs in each case.

-- A proposed minimum net bottom clearance. On 5 May 1976, the Coast Guard published in the Federal Register a proposed policy that there be a stated minimum net clearance between the hulls of vessels and the bottoms of the waterways. If a policy for a mininum net clearance is implemented by the Coast Guard it could result in a significant increase in the total annual dredging workload for hopper dredges. Such a program would cause an increase in the dredging frequency cycles as well as an increase in the volume of material to be removed from the waterways. As a result, additional hopper dredges beyond those currently envisioned could be required.

-- An increasing trend in the usage of hopper dredges on beach nourishment and hurricane protection projects. Since 1966 there has been a continuing increase in the use of hopper dredges on these types of projects. This trend is expected to increase in the future due to the need to obtain the materials for beach nourishment and hurricane protection projects from borrow areas located in the offshore zone rather than from sources located within the estuaries. This change in the source of dredged materials results from environmental restrictions and constraints on the excavation of materials from the estuaries. The use of hopper dredges for beach nourishment and hurricane protection projects involves operations in exposed waters and frequently involves the discharge of the excavated materials through long discharge pipelines. This new and extensive requirement which involves a need for an emergency response capability requires a large percentage of dredge production time that would otherwise be available to provide for emergencies on navigation projects.

-- It is impossible to predict the effects of industry strikes and contractural commitments, including domestic and overseas requirements, on the availability of industry dredges. However, it has been assumed that industry strikes and contractural commitments to other than Corps of Engineers projects will be regional in nature and limited in scope.

-- The Federal Water Pollution Control Act (PL 92-500, 18 October 1972) known as the Clean Water Act and the Marine Protection, Research, Sanctuaries Act (PL 92-532, 23 October 1972) known as the Ocean Dumping Act, have in recent years caused an increase in the dredge production time required on navigation projects. Under the provisions of these laws dredged materials are now classified as polluted or unpolluted by the Environmental Protection Agency. These laws have caused the hopper dredge production time at the various projects to increase due to two factors. First, it is necessary in most cases to haul the dredged materials a greater distance to open water disposal sites than in the past. Secondly, the requirement in some cases to place the dredged materials in diked disposal areas increases the normal cycle time by a factor of two to four. The recent delegation of the responsibility for the issuance of permits for disposal areas to the states is expected to lead to further increases in haul distances and in the usage of diked disposal areas.

-- Conference Report No. 95-1490, dated August 14, 1978, on the fiscal year 1979 Appropriation Act indicates that the Congress is concerned over the adverse impact that the implementation of the Seagoing Barge Act provisions would have if they are applied to the dredging operations of the nation. It is estimated that the vast majority of the industry cutterhead dredges which have operated in the offshore zone on beach nourishment and hurricane protection projects will not qualify for operations in the offshore zone under a strict interpretation of the provisions of the Seagoing Barge Act (46 USC 395). If the Coast Guard proceeds with an immediate and strict implementation of the provisions of the Act, and it seems very likely that this will be the case, it will result in the need for additional seagoing hopper dredges to perform the substantial volume of work previously performed with cutterhead dredges. The impact of this action could result in a drastic change in the requirement for hopper dredges.

Status of the Study

The U.S. Army Corps of Engineers study on the minimum hopper dredge fleet requirements was forwarded to the Secretary of the Army by the Chief of Engineers on 1 February 1979. The Secretary of the Army approved the Chief of Engineers recommendation that the minimum hopper dredge fleet should include eight hopper dredges and forwarded the study to the Office of Management and Budget (OMB) on 6 February 1979. The study is under review by the OMB staff and is expected to be forwarded to the Congress well in advance of the April 26, 1980 deadline established by the Congress in Public Law 95-269.

Risk Factors Included in the Study

The Civil Works Directorate assessment that eight hopper dredges are needed in the minimum fleet of the Corps of Engineers accepts the following risk factors:

-- That eight hopper dredges will provide the necessary production capability to meet either the overseas or the continental U.S. defense requirements, but is not sufficient to provide for both requirements at the same time.

-- That emergency dredging requirements will not occur on a broad scale or in more than one region at a time.

-- That vessel losses due to sinkings will not occur.

-- That the requirements for overseas commitments will be limited in scope and will not occur frequently.

-- That industry strikes will occur infrequently and only on a regional basis.

-- That the usage of industry hopper dredges outside the United States and/or their usage under contractural commitments within the continental United States would be limited and for short duration.

-- That extended delays due to groundings, weather and sea conditions will not occur.

-- That vessel losses or extended lost time due to enemy action will not occur.

-- That damages and lost time as a result of collisions and other hazards of dredging operations will not occur.

-- That extended delays due to major mechanical problems will not occur.

-- That emergency and national defense requirements will not occur during a period when several dredges are incapacitated simultaneously by scheduled and unscheduled shipyard repairs.

-- That repairs of a vessel deployed to address a national defense requirement would not be required for at least six months from the date of deployment.

CONCLUSIONS

-- That the level of annual funding since 1963 has decreased significantly and led to the industry experiencing financial problems.

-- That Public Law 95-269 provides for the Corps of Engineers to maintain a minimum fleet of technologically modern dredges in a fully operational status to respond immediately to emergency conditions, national defense and national interest requirements both in the United States and abroad.

-- That the industry is responding to the opportunity to compete with Corps plant for additional dredging work under the Industry Capability Program.

-- That the industry will construct additional modern hopper dredges.

-- That the majority of the existing Corps of Engineers hopper dredges are obsolete and must be retired.

-- That the retirement of the existing obsolete Corps hopper dredges must be carefully analyzed and equated to a clear indication that the industry hopper dredges are capable of extended efficient performance.

-- That eight hopper dredges are needed in the minimum Federally-owned fleet: one large class, four medium class and three small class hopper dredges.

-- That a request for funds to construct additional hopper dredges will not be made until fiscal year 1983 to allow the industry additional time to compete with Corps dredges.

SEA BOTTOM MANAGEMENT IN JAPAN

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INTRODUCTION

Japan is surrounded by the sea and most of her people live in coastal areas. "Cleaning of the sea" is one of the urgent tasks facing the government and the people of Japan. Although we had highly accelerated economic growth in the 1960s, the environmental problems did not attract concern. However, in the 1970s, environmental pollution progressed to the point of adversely affecting our way of life. Now, the need for environmental preservation of the sea and ports and harbors has attracted the attention of the nation. To cope with the need for environmental improvement of ports and harbors, extensive administrative measures have been taken. Projects that have been implemented to upgrade the environment of ports and harbors under central government authority are shown in Table 1. There are two types of projects involving disposal and treatment of polluted bottom sediments. One is to take preventive measures against water pollution in the ports and harbors. This work is directed primarily toward the removal of polluted bottom sediments. The other aspect includes studies on the feasibility of purification of organic sediments that have accumulated and are widely distributed in the bays and inland seas.

This paper will discuss some of the projects, now underway or planned as Improvement Works for ports facilities.

SEA BOTTOM CLEAN UP PROJECTS IN PORTS AND HARBORS

Preventive Measures

The Federal Law relating to Special Measures on Finance for Pollution Prevention was enacted in May 1971. Under this law, the Japanese government subsidizes dredging or water-intake projects authorized in the Pollution Control Program. This public waters Pollution Control Program has been established in 50 areas, including Yokkaichi, Tokyo and Kitakyushu. Where the program is not established, projects must receive special designation by the Minister of Home Affairs in order to obtain central government funding.

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TABLE 1. ENVIRONMENT IMPROVEMENT PROJECTS FOR PORTS AND HARBORS

Projects	Law Applicable	Ratio of Burden of Subsidy	Components to be Subsidized	Outline of the Systems
Construction of facilities to protect water from oil pollution	The Law relating to Prevention of Marine Pollution and Maritime Disaster	5:10	Construction of waste oil plant or its improvement	Construction or improvement of the ship dumping oil disposal facilities
Pollution prevention measures in ports and harbors	The Law relating to Special Measures on Finance by Central Government concerned with Projects for Pollution	1:2	Dredging, water intaking, etc.	Polluted bottom deposit dredging operation or water intaking works designated in the pollution control program or designated by the Minister of Home Affairs
Construction of waste disposal facilities, etc.				
Construction of revetments to contain dredged materials	Port and Harbor Law	2.5:10	Construction or improvement of the revetments for disposal area	Construction or improvement of the revetments for disposal area
Construction of the marine waste disposal facilities (receiver, incinerator, crasher)	Port and Harbor Law, The Law relating to Special Measures on Finance by Central Government concerned with Projects for Pollution Prevention	2.5:10 (5:10)	Construction of the marine waste disposal facilities	Construction of these facilities designed to treat marine wastes discharged from the ship or the facilities at sea
Construction of a port cleaning ship	Subsidization on Finance	2.5:10	Construction of port cleaning ship	Construction of the ship designed for port cleaning
Disposal of wrecked ship	Subsidization on Finance	3:10	Disposal of abandoned ship	Disposal of abandoned ship
Stockpiling materials and supplies to protect ports and harbors from pollution	Subsidization on Finance	2.5:10	Stockpiling of oil fence	Stockpiling of the oil fence for environmental conservation in the ports and harbors
Building the green zone	Port and Harbor Law	1:2 (for facilities) 1:3 (for reclaimed land, etc.)	Building the green zone	Building or improving the seashore green zone, the park and rest house, etc.
Marine environment improvement	Financial Measure	10:10	Sea cleanup operation	Collection of the floating oil or garbage
Survey for the project	Financial Measure	10:10	Survey	The survey for execution of the project

When there are businesses or industries that create pollution, those companies must share the pollution control or clean-up costs in proportion to their contribution to the problem. Table 2 outlines pollution prevention measures in ports and harbors now underway or planned.

Effects of Project Implementation

Clean-up procedures for polluted bottom sediments can be divided into two categories according to the nature of the pollutant--one is used for bottom deposits containing organic substances and the other process is to contain toxic substances. The effectiveness of the latter can be checked by measuring amounts of toxic substances contained in the sediments after the process is complete. Accumulation of toxic substances in marine animals is measured also.

When bottom sediments are contaminated by organic substances, sediment removal is effective. Effectiveness can be evaluated in terms of improvement in water quality near the dredging site and the reduction in odor.

In this paper, two bottom sediment clean-up projects are discussed. One is the dredging operation in Tokuyama-Kudamatsu port and the results of environment surveys before and after the operation. The other is the project at Hachinohe port.

Project involving sediment contaminated by toxic substances

Tokuyama-Kudamatsu port in Tokuyama Bay, in the western part of the Seto Inland Sea, was heavily polluted by mercury (see Figure 1). The Japanese tentative standard for mercury is set at 15 ppm; therefore the area to be cleaned was confined to that where the mercury in the sediments exceeded the standard. Dredging had occurred at the eastern site from June, 1975 to November, 1976 and, at the western site from September, 1975 to March, 1977. Two kinds of operations were used. One was the dredging of polluted sediments and the other involved containing the polluted dredged materials. The disposal site foundation was improved with the sand drain and sand compaction pile method. After the foundation was reinforced, two rows of sheet pile and sheet pile cellular cofferdam were installed. During the filling with dredged materials and treatment of effluent water, precautions were taken to avoid secondary pollution. No abnormal values of environmental parameters were found during the project. After completion, the bottom sediment, water quality and fish were examined (see Figure 2, Table 3).

Examination of bottom sediment. To assure removal of the sediments containing excessive amounts of mercury, analyses were made of samples taken around the project site. Results (Table 4) indicate the project effectiveness; no bottom sediments containing over 15 ppm mercury were found. No dispersion of the dredged materials occurred in the area.

Examination of the fish. Five species of fish that had been affected by mercury poisoning have been checked annually since 1974. As can be seen in Table 5, mercury in fish tissues was reduced to a permissible limit after the dredging operation.

TABLE 2. OVERVIEW OF PROJECTS INVOLVING REMOVAL OF BOTTOM SEDIMENT AT PORTS AND HARBORS (as of March 1979)

Name of Port	Work Period	Pollutant	Spoil x 1000 m^3	Removal Standard	Reclamation Method
Tokyo	1972-1981	organics	2,400	6 points or more in the overall, ignition loss, COD, and sulfides	Reclamation by sealed grab bucket and special pumps
Yokohama	1973-1979	organics	691	same as Tokyo	Reclamation by ordinary grab buckets
Nagoya	1972-1981	mercury, organics	729	Hg, 25 ppm; ignition loss, 15%	Reclamation by ordinary grab buckets
Yokkaichi	1974-1979	mercury, oils	2,200	Hg, 6 ppm; oils, 4,000 ppm	Reclamation by special pumps
Osaka	1973-1981	organics	1,645	ignition loss, 15%	Reclamation by special pumps
Himeji	1974-1980	organics	460	ignition loss, 15%	Dredging by ordinary pump, solidification and reclamation
Wakayama Shimotsu	1979-1980	organics	100	pending decision on either COD 20 mg/g or sulfides 1 mg/g	
Kita-Kyushu	1972-1981	mercury organics	3,300	Hg, 30 ppm	Reclamation by sealed grab buckets
Shiogama	1972-1977	organics	47	ignition loss, 15%	Reclamation by sealed grab buckets
Tagonoura	1972-1980	PCB, organics	1,720	PCB 10 ppm	(1) Reclamation by pump, (2) Dredging by ordinary grab buckets, solidification, and reclamation
Mizushima	1972-1980	oil	813	1,500 ppm	
Minamata	1975-1984	mercury	1,675	Hg, 25 ppm; COD, 30 mg/g	
Hachinohe	1979-1980	organics	257	sulfide, 1 mg/g; ignition loss, 10%	Reclamation by pump
Amagasaki, Nishinomiya, Ashiya	1977-1981	organics	200	ignition loss, 15%; COD, 20 mg/g or sulfides, 1 mg/g	Dredging by ordinary grab buckets, solidification, reclamation
Higashi-Hirama	1978-1980	organics	50	undecided	
Kure	1980	organics	200	undecided	
Takamatsu	1980	PDB, organics	397	undecided	
Toyo	1977	organics	20	odor no., 2.5	Reclamation by sealed grab buckets
Iwakuni	1980	organics	372	undecided	
Mitajiri-Nakanoseki	1976-1978	organics	445	ignition loss, 15%	Reclamation by special pumps
Ube	1980	organics	492	undecided	
Saeki	1980-1981	organics	805	undecided	

(continued)

TABLE 2. (continued)

Name of Port	Work Period	Pollutant	Spoil x 1000 m^3	Removal Standard	Reclamation Method
Otake	1980	organics	890	undecided	
Omuta	1973-1979	cadmium, mercury	536	Hg, 25 ppm; Cd, 200 ppm	(1) Reclamation by special pumps; (2) Earth covering
Nakatsu	1980	organics	1,008	undecided	Earth covering
Rumoi	1980	organics	15	undecided	
Matsuyama	1973-1974	organics	62	odor no., 2.5	Reclamation by sealed grab buckets
Mikawa	1973-1975	organics	50	COD, 16 mg/g; ignition loss, 10%; sulfides, 1 mg/g	Reclamation by sealed grab buckets
Sakata	1974-1975	mercury	71	Hg, 28 ppm	Reclamation by special pumps
Aburatsu	1974	organics	18	ignition loss, 20%	Reclamation by ordinary grab buckets

TABLE 3. OUTLINE OF BOTTOM CLEAN-UP PROJECT AT PORT OF TOKUYAMA-KUDAMATSU

		Eastern Site	Western Site
Work Period	Start	July 7, 1975	September 4, 1975
	End	November 11, 1976	March 15, 1977
Disposal Site	Area (m^2)	29,000	451,000
	Capacity (m^3)	365,000	5,479,000
Dredging	Area (m^2)	239,000	294,000
	Volume (m^3)	148,000	214,000
Polluted Area (m^2)		145,000	451,000

TABLE 4. RESULTS OF THE BOTTOM SEDIMENT ANALYSES

Site		Number of Points	Total Mercury (mg/kg) Average	Total Mercury (mg/kg) Range
Eastern Site	T - 1	6	2.2	0.26 - 3.8
	T - 2	5	1.7	0.03 - 2.7
	T - 3	9	1.2	0.14 - 4.0
	T - 4	20	2.3	0.08 - 7.6
	T - 5	9	2.9	0.95 - 5.2
	T - 6	7	3.1	0.71 - 8.4
	Total	56	2.3	0.03 - 8.4
Western Site	N - 1	14	1.9	0.03 - 7.7
	N - 2	12	2.6	0.14 - 8.5
	N - 3	12	2.4	0.13 - 5.0
	N - 4	12	3.3	0.56 - 6.5
	N - 5	12	2.9	0.19 - 7.5
	Total	62	2.6	0.03 - 8.5

TABLE 5. CONCENTRATION OF ACCUMULATED MERCURY IN FISH FROM TOKUYAMA BAY

	Sept 1973 -Oct 1975	June 1976	Nov-Dec 1976	Jun-Jul 1977	Oct-Nov 1977	June 1978	Oct-Nov 1978	Jun-Jul 1979
Rock Trout	(165)* 0.629†	(30) 0.322	(10) 0.303	(10) 0.273	(15) 0.206	(20) 0.176	(25) 0.208	(25) 0.179
Japanese Sea Perch	(195) 0.428	(30) 0.315	(20) 0.178	(20) 0.199	(30) 0.199	(25) 0.210	(20) 0.240	(30) 0.183
Sea Bass	(145) 0.416	(10) 0.487	(25) 0.130	(20) 0.215	(15) 0.249	(25) 0.203	(20) 0.216	(20) 0.204
Rock Fish	(110) 0.529	(25) 0.460	(30) 0.217	(30) 0.246	(15) 0.264	(25) 0.257	(20) 0.260	(30) 0.158
Black Porgy	(115) 0.639	(20) 0.601	(20) 0.425	(25) 0.545	(30) 0.398	(30) 0.423	(25) 0.357	(25) 0.349
Average	(730) 0.528	(115) 0.437	(105) 0.250	(105) 0.295	(105) 0.263	(125) 0.253	(110) 0.256	(130) 0.214

* Number of samples (n). † Total mercury (mg/kg).

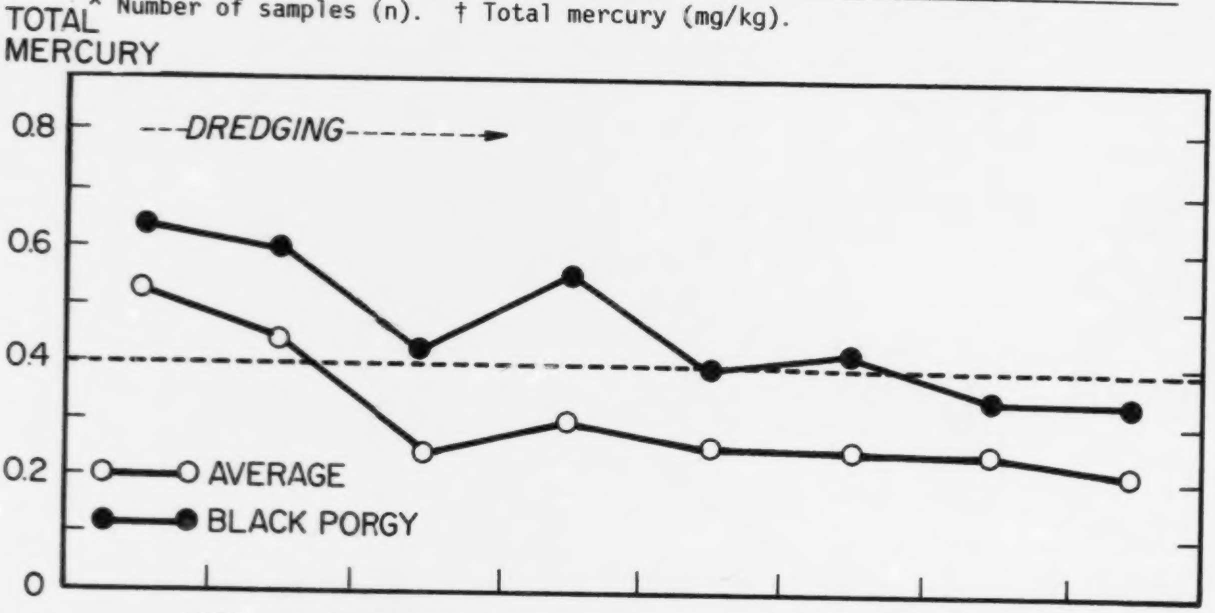

TABLE 6. PROPERTIES OF BOTTOM SEDIMENT IN PORT OF HACHINOHE*

							Surface Layer	
	COD	Sulfide	Ignition Loss	T-N	T-P	Cd	Pb	Total Mercury
Property	39.5 mg/g	5.5 mg/g	12.5%	3.2 mg/g	2.7 mg/g	5.3 μg/g	311 μg/g	12.0 μg/g

* Average of 1973-1978.

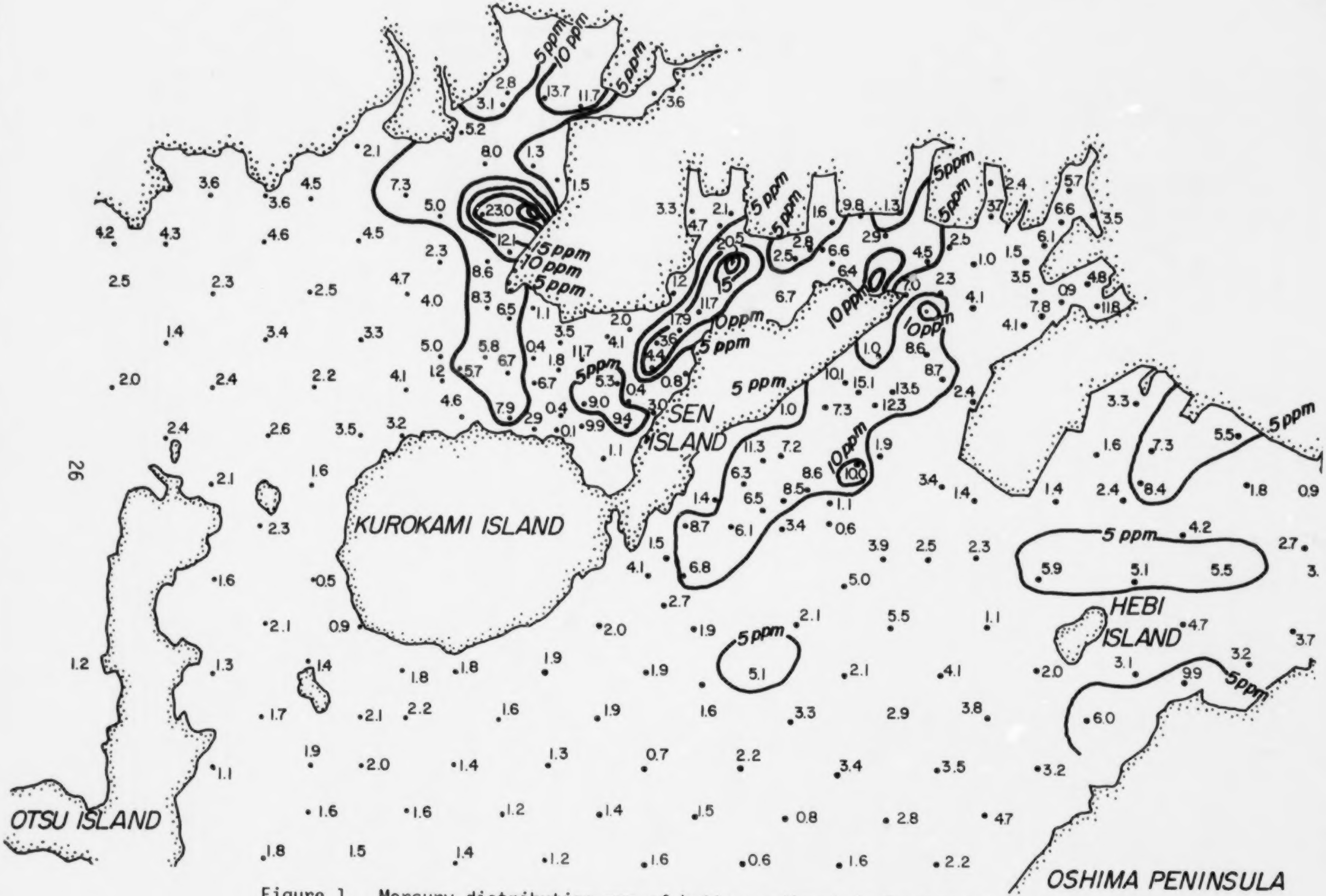

Figure 1. Mercury distribution map of bottom sediment in Tokuyama Bay (mg/g).

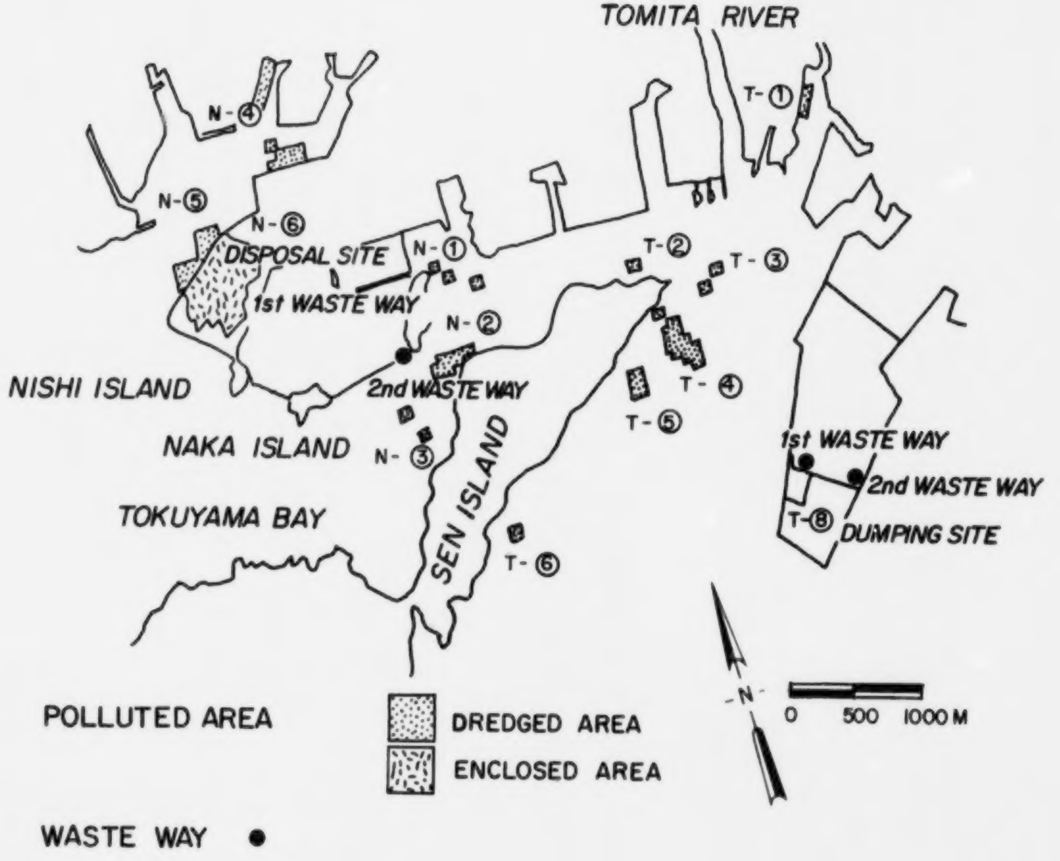

Figure 2. Dredged or enclosed area in Port of Tokuyama-Kudamatsu.

Project involving sediment contaminated by organic substances

This project took place in the Hachinohe Port in Aomori prefecture. Deep within the port is an industrial site where the water is always stagnant. In 1978 COD measured 7 mg/l, indicating extensive water pollution. Organic deposits (see Table 6) are apparent sources of water pollution and offensive odors. To eliminate these pollution sources, certain standards for removal of organic deposits (Table 7) were set, providing the basis for the "clean water" project of Hachinohe Port. The project duration is scheduled from 1979 to 1980. During the operation, as much as 256,800 m^3 of bottom deposits will be dredged and reclaimed as land fill material for the green area project (refer to Figure 3).

The example of Hachinohe Port is appropriate to assess the effects of clean-up since the project period is relatively short and the topography is suitable for this purpose. Site surveys before and after the project will measure the degree of pollution and, at the same time, provide data for an analytical study of the mechanism and function of the pollutants contained in bottom sediments. Main items of the surveys are listed in Table 8. Plans include studies to determine the critical mass of organic substances causing environmental pollution and increase the understanding of mechanisms by which organic substances are released into the water and the causes of offensive odors.

BOTTOM SEDIMENT CLEANUP IN BAYS AND INLAND SEAS

Countermeasures for Pollution of Bottom Sediment

Tokyo Bay, Ise Bay and the Seto Inland Sea are confined water bodies. There are many sources of pollution and the water exchange dynamics are poor, resulting in extensive pollution problems. Thus, it is essential to take measures to improve water quality in these areas. Pollutants contained in domestic and industrial liquid wastes have settled to the seabed and, over many years, formed the highly polluted deposits. These deposits have contaminated the marine waters through release of organic substances and nutrient salts; offensive odors and oxygen deficiency are associated with the release of organic substances from bottom sediment.

The present state of bottom sediment contamination in Tokyo Bay, Ise Bay and the Seto Inland Sea is illustrated in Figure 4. It is apparent that eutrophication and deterioration of the self-purification capacity of the waters can be attributed to the polluted deposits. In this situation extensive bottom cleanup projects are planned to improve the marine environment. Basic surveys have been conducted on a wide scale for Tokyo Bay, Ise Bay and the Seto Inland Sea. Through the survey results, the benefits of the bottom cleanup projects became apparent. However, such a cleanup operation encompassing this vast area is no easy task. Before the project can be implemented it is necessary:

1) To identify priority programs for the project.
2) To perfect sediment removal methods.
3) To make preliminary studies of potential impacts on the environment.

TABLE 7. PROVISIONAL REMOVAL STANDARD OF BOTTOM SEDIMENTS, HACHINOHE

	Removal Standard
COD	30 mg/g
Sulfide	1 mg/g
Ignition Loss	10%

TABLE 8. MAJOR PARAMETERS SURVEYED IN HACHINOHE PORT

Parameter		Number of Sampling Sites	Items	Remarks
Current		3		24 Hours Observation
Water Quality		5	pH, Cl^-, COD, DO, TOC, T-P, PO_4-P, T-N, NH_4-N, SS, chlorophyll a, water color, n-hexane extraction substance, etc.	
Bottom Sediments	Content	2	ignition loss, COD, sulfide, T-N, T-P, TOC, Eh, n-hexane extraction substance, Fe^{2+}, Mn^{2+}, etc.	
	Void Water	2	COD, T-P, PO_4-P, T-N, NH_4-N, Eh, Fe^{2+}, Mn^{2+}, etc.	
	Release	1	COD, TOC, T-N, NH_4-N, T-P, PO_4-P, etc.	
	Oxygen Demand	1	DO, S^{2-}, etc.	
Organism		5	benthos, plankton, adnate organisms, bacterium, etc.	
Odor		3	H_2S, CH_4, CH_3SH, etc.	

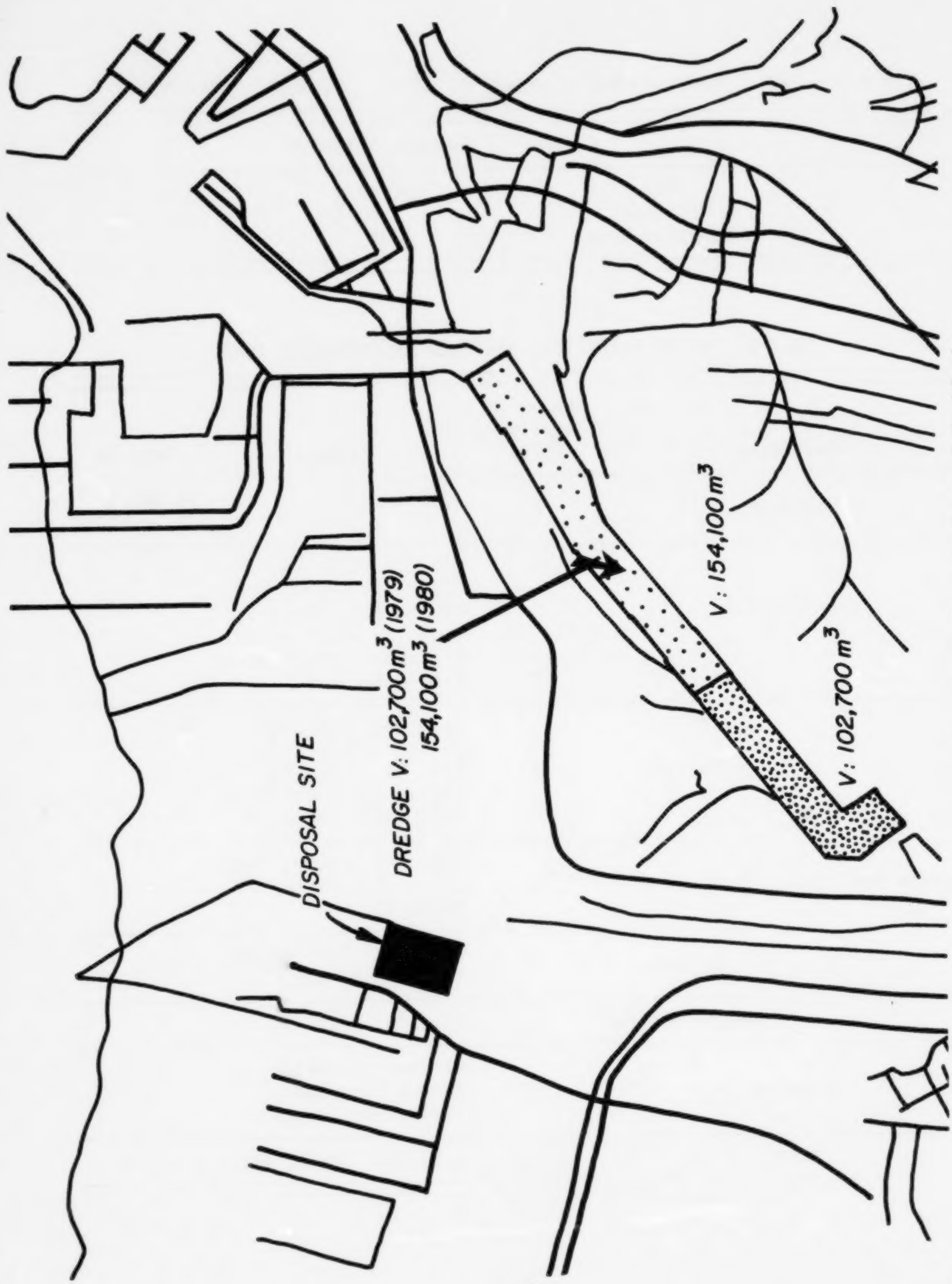

Figure 3. Sediment cleanup project in Port of Hachinohe.

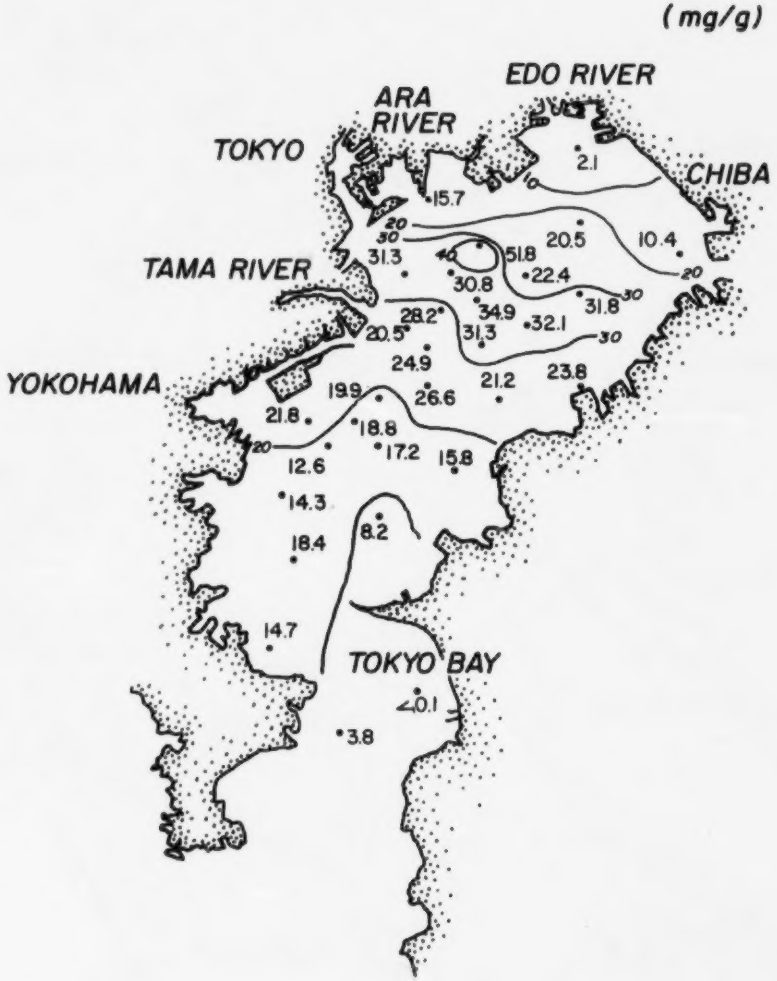

Figure 4a. Horizontal distribution of COD (sediment) in Tokyo Bay, September 1977 (mg/g).

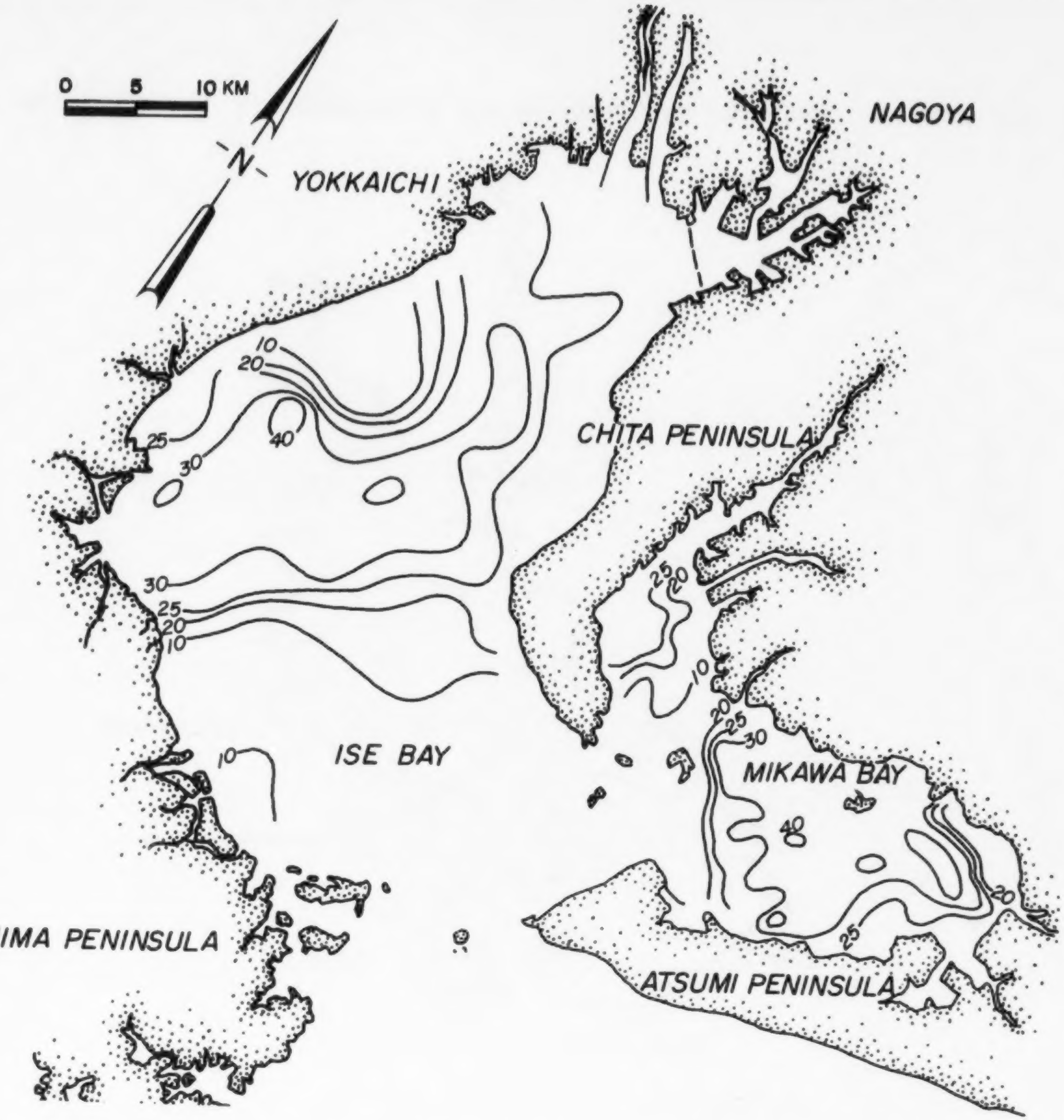

Figure 4b. Horizontal distribution of COD (sediment) in Ise Bay, 1975 (mg/g)

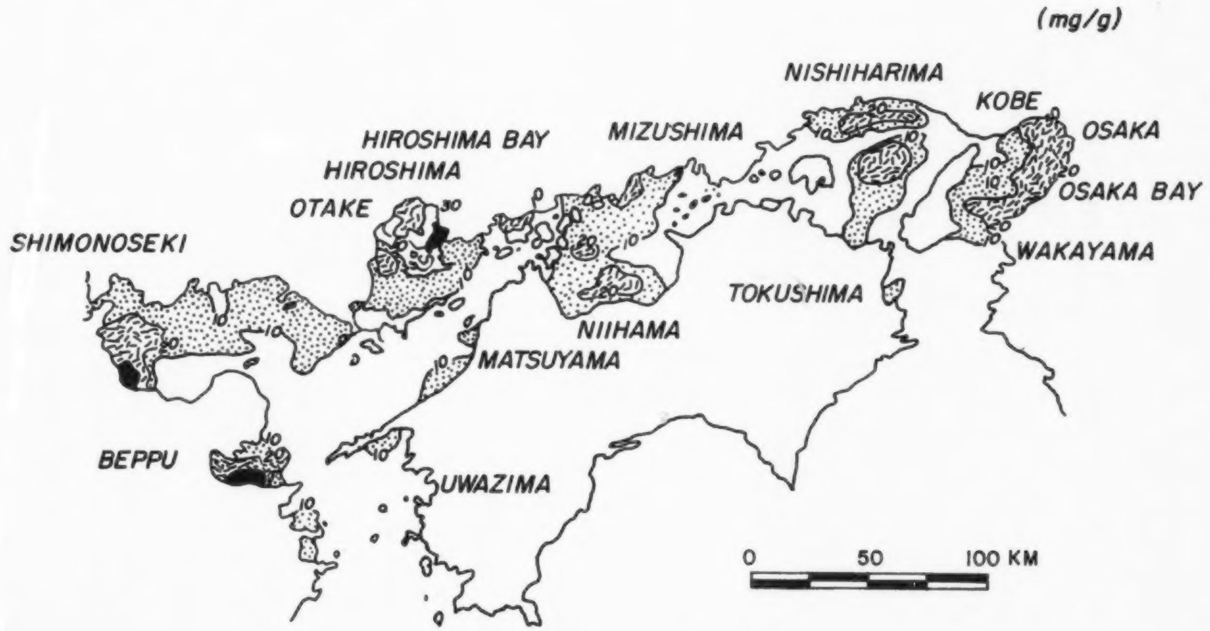

Figure 4c. Horizontal distribution of COD (sediment) in Seto Inland Sea, August 1972 (mg/g)

Therefore, before initiating the main operation, a pilot project should be designed. The pilot project will serve as the model for most contaminated sea bottom operations. The survey began in 1979 in the Seto Inland Sea with the following objectives:

1) To study the mechanics of sea sediment pollution.
2) To analyze sediment pollution and the effects of removal.
3) To implement the experiments.
4) To study the impact of the project on the environment.

Effects of Bottom Cleanup in Bays and Inland Seas

One of the most important objectives of the pilot project is the analysis of sea water pollution mechanisms and the effects of bottom cleanup. Because of cost and time limits, cleanup in large areas such as the Seto Inland Sea must be efficient. Therefore it is essential to the overall project that there is an understanding of the pollution mechanisms and the ways polluted deposits can affect their aquatic environments.

The plan calls for the development of a model of the mechanisms which cause pollution and for analytical studies on the effects of the cleanup. The model is being designed with data from related research projects, site surveys and experimental demonstrations.

Concept of the circulating system of organic substances

It seems that water pollution progresses through a circulating system (generation-dissolution-transfer-dispersion) in the Seto Inland Sea. Therefore the model is being developed based on the concept of the circulating system of organic substances (Figure 5).

Assessments of the model

Assessments of the improved water quality after cleanup should be made on the primary (direct) effect and secondary (indirect) effect, taking into consideration the time (season) and space (place) factors. The primary effect is that attributed to removal of substances which would be released from bottom sediments; secondary effects are those influencing oxygen demand and the release of inorganic nutrient salts (phosphorus). The suppression of inorganic nutrient salt (phosphorus) release will retard the development of eutrophic conditions in surface waters.

Functions of the model

(1) To model the cyclic circulation mechanisms of the pollutants between water and bottom sediments.

(2) To provide a fit for seasonal variation and yet be reliable for long term observations.

(3) To provide a model that can predict the probable effects on water quality and simulate the effects of bottom cleanup.

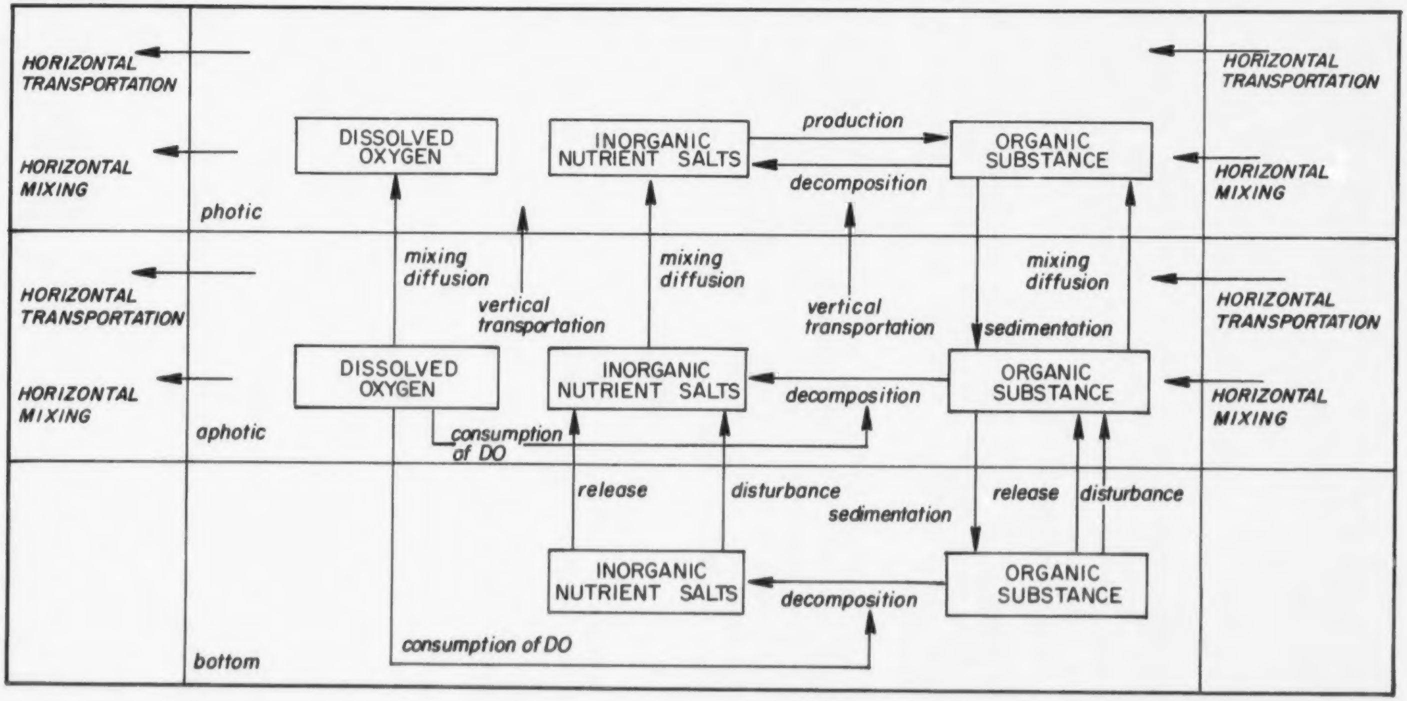

Figure 5. Concept of the circulating system of the organic substances.

(4) To model the substances' cycling mechanisms (generation, dissolution, sedimentation and release) and the relationship between dissolved oxygen and releasing speed.

FUTURE PROSPECTS

Federal government subsidized cleanup operations in the ports and harbors beginning in 1972. Since then, some of the technical problems have been solved through the actual projects or by conducting research studies and surveys. Today's most critical problem is to establish a realistic standard for sea bottom sediment cleanup and to adopt more effective cleanup methods.

Full scale surveys of the organically polluted bottom deposits in bays and inland seas have just begun. Therefore, based on the relevant available data, a comprehensive program of research and implementation must be established.

Currently there is the problem of eutrophication in these waters. The national projects of sea bottom cleanup combined with pollutant-loaded effluent control programs are attracting considerable attention. The objectives of this project lie not only in the removal of years' accumulation of pollutants, but also in the creation of a better environment and ultimate improvement of the coastal waters.

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CONTROL OF TOXICS IN THE UNITED STATES

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INTRODUCTION

The problems associated with disposal of toxic substances in the United States have been spotlighted in the past few years for two primary reasons. First, the Toxic Substances Control Act of 1976 (TSCA) gave the government broad new authority to gather information on the potential of chemicals to damage human health and the environment and authority to control these substances where necessary. Second, an even greater awareness of chemical dangers has resulted from a recent series of unrelated toxic disposal episodes in the United States.

The control of toxic substances is extremely complex as indicated by the number of different chemicals produced, the number of manufacturers, and the amounts of chemicals produced. In November 1977, the registry of chemicals maintained by the American Chemical Society listed 4,039,907 distinct chemical compounds—and the registry includes only chemicals reported in the literature since 1965. The list has been growing at a rate of 6,000 per week (1). Chemicals currently in commercial production in the U.S. may number as high as 70,000; 50 are produced in quantities greater than 1.3 billion pounds per year (2). From 115,000 companies involved in the production and distribution of chemicals, the business is worth $113 billion per year, or about 7 percent of the nation's GNP (3). The majority of these chemicals probably are innocuous and beneficial, yet our knowledge of them is limited.

TYPES OF PROBLEMS

Increasingly, the hazards of toxic substances—real or suspected—in the home, marketplace and workplace are being reported in the popular as well as the technical literature. These substances reputedly permeate the air we breathe, the water we drink and the food we eat. Examples of these problem types are:

- PCB (polychlorinated biphenyl) contamination of food is appearing across the nation. At least nine states have restricted the consumption of fish caught in local waters because concentrations of PCBs exceed the U.S. Food and Drug Administration allowable limits (4). Accidental contamination of turkey feed with PCBs in the northwestern U.S. has resulted in withdrawal of thousands of birds from the marketplace. Public concern has resulted in a precipitous decline in the sale of uncontaminated turkeys with the secondary effect of seriously damaging the turkey industry. (PCBs are now banned from sale in the U.S. but large amounts are already in the environment or contained in electric capacitors and transformers which present a serious disposal problem).

- Recent findings have shown people living near roadways may face up to nine times the normal chance of contracting cancer due to their exposure to polycyclic aromatic hydrocarbons found in automobile exhaust (5).
- Evidence shows that florists and others working with large quantities of cut flowers are exposed to dangerous levels of toxics from the pesticide or chemical residues on the plants (6).
- Benzene is found in many consumer products such as paint strippers, carburetor cleaners, denatured alcohol, rubber cement and art and craft supplies. Studies show that levels in households frequently exceed standards allowed for occupational exposure. Excessive benzene exposure is suspected of inducing leukemia (7).
- Across the nation old industrial waste disposal sites are being viewed with alarm due to the recent discovery of highly toxic materials in New Jersey, New York and Kentucky. In the Love Canal area of Niagara Falls, New York where homes and a school were built over an old chemical waste disposal site containing rusting barrels of more than 82 chemicals, 11 of them suspected carcinogens, there has been an unusually high incidence of birth defects and miscarriages. The seriousness of the disposal site problem was reinforced by a recent Congressional report which concludes "even an extraordinary effort commenced immediately, cannot achieve adequate protection for the American public for years to come" (8).
- According to a survey of 4,636 U.S. industrial plants employing a total of 985,000, one of four citizens is exposed in the workplace to some substance capable of causing death or disease (9).

FEDERAL LEGISLATION

It is obvious that the possibilities for involuntary exposure to toxic substances are almost endless. Government and industry, together, are responsible for protecting workers, the public and the environment. A series of federal laws have been enacted to protect the public health and the environment from toxic substances (see Table 1). The focus of these laws parallels the routes of involuntary exposure, e.g. air, water, food, drugs, etc. Toxic laws are primarily extensions of public health and environmental protection efforts. The scope of these laws and control philosophies has changed over time.

In the early 1970s environmental legislation focused on controlling pollution in specific media (air, water, land) after the pollutant had been generated. The methods included setting "ambient" air standards or "receiving water" standards and specific emission or discharge standards to limit the amount of a pollutant allowed into the environment. More recent legislation supports a preventive philosophy. For example, the Toxic Substances Control Act and recent amendments to the Federal Insecticide, Fungicide and Rodenticide Act require chemicals to be tested before they can be manufactured for commercial use. The evolving toxics control philosophy in the United States emphasizes the need to protect total ecological systems and to integrate the regulation of industrial discharges to insure that the requirements for air, water and solid waste discharges are compatible.

The diversity of toxics-related laws is illustrated in Table 1. The most comprehensive of these are administrated by the Environmental Protection Agency (EPA). The Toxic Substances Control Act (TSCA) and the Resource Conservation and Recovery Act (RCRA) are the most significant in terms of control authority. TSCA requires premarket toxicological testing of all new chemicals. All existing chemicals in commerce—excluding pesticides—must be identified and testing can also be required. EPA has been given broad authority to ban, limit, or modify the use, manufacture, or processing of any substance which could pose an unreasonable risk to human health or the environment. (Details of TCSA were presented by E. Wallen at the Third U.S.-Japan Experts' Meeting in 1978.) RCRA establishes a comprehensive government regulatory system for control of hazardous wastes from generation to ultimate disposal. It prohibits open dumps and the disposal of hazardous wastes in sanitary land fills. RCRA also requires criteria to be established for construction of proper disposal facilities and encourages recovery and recycling activities.

Following is a summary of other important federal laws dealing with the regulation of toxics:

- Clean Air Act (as amended 1970, 1977) regulates emissions from both mobile and stationary sources and sets ambient standards for major classes of pollution based primarily on public health considerations. It also controls point source emissions of "hazardous air pollutants."
- Federal Water Pollution Control Act (as amended 1972, 1977) sets forth procedures for establishing and enforcing water quality standards for receiving waters and effluent standards for point source discharges of pollutants. Provides for control of toxic pollutants in effluents and enforcement in toxic spill events. Requires that, by 1984, best available treatment technology be applied to 65 classes of toxic chemicals for 21 industries and provides for cleanup action against other potentially toxic chemicals by 1987.
- Occupational Safety and Health Act (1970) sets exposure standards for toxic and hazardous materials in the workplace so that "no employee will suffer material impairment of health or functional capacity."
- Both the Consumer Protection and Safety Act (1972) and the Federal Hazardous Substances Act (as amended 1960, 1969) give broad power to limit or prevent public exposure to toxic or hazardous materials in consumer products (excluding tobacco, foods, drugs and cosmetics).

TABLE 1. FEDERAL LAWS AND AGENCIES AFFECTING TOXIC SUBSTANCES CONTROL*

Statute	Year Enacted	Responsible Agency	Sources Covered
Toxic Substances Control Act	1976	EPA	Requires premarket evaluation of all new chemicals (other than food additives, drugs, pesticides, alcohol, tobacco); allows EPA to regulate existing chemical hazards not covered by other laws related to toxic substances
Clean Air Act	1970, amended 1977	EPA	Hazardous air pollutants
Federal Water Pollution Control Act	1972, amended 1977	EPA	Toxic water pollutants
Safe Drinking Water Act	1974, amended 1977	EPA	Drinking water contaminants
Federal Insecticide, Fungicide, and Rodenticide Act	1948, amended 1972, 1975, 1978	EPA	Pesticides
Act of July 22, 1954 (codified as §346(a) of the Food, Drug and Cosmetic Act	1954, amended 1972	EPA	Tolerances for pesticide residues in food
Resource Conservation and Recovery Act	1976	EPA	Hazardous wastes
Marine Protection, Research and Sanctuaries Act	1972	EPA	Ocean dumping
Food, Drug and Cosmetic Act	1938	FDA	Basic coverage of food, drugs and cosmetics
Food additives amendment	1958	FDA	Food additives
Color additive amendments	1960	FDA	Color additives
New drug amendments	1962	FDA	Drugs
New animal drug amendments	1968	FDA	Animal drugs and feed additives
Medical device amendments	1976	FDA	Medical devices
Wholesome Meat Act	1967	USDA	Food, feed and color additives and pesticide residues in meat and poultry
Wholesome Poultry Products Act	1968		
Occupational Safety and Health Act	1970	OSHA	Workplace toxic chemicals
Federal Hazardous Substances Act	1966	CPSC	"Toxic" household products (equivalent to consumer products)
Consumer Product Safety Act	1972	CPSC	Dangerous consumer products
Poison Prevention Packaging Act	1970	CPSC	Packing of dangerous children's products
Lead Based Paint Poison Prevention Act	1973, amended 1976	CPSC	Use of lead paint in federally assisted housing
Hazardous Materials Transportion Act	1970	DOT (Materials Transportation Bureau)	Transportation of toxic substances generally
Federal Railroad Safety Act	1970	DOT (Federal Railroad Admin)	Railroad safety
Ports and Waterways Safety Act	1972	DOT (Coast Guard)	Shipment of toxic materials by water
Dangerous Cargo Act	1952		

CPSC = Consumers Product Safety Commission
DOT = U.S. Department of Transportation
EPA = U.S. Environmental Protection Agency
FDA = Federal Drug Administration
OSHA = Occupational Safety and Health Administration
USDA = U.S. Department of Agriculture

* From "Environmental Quality," the Ninth Annual Report of the Council on Environmental Quality, December 1978.

- Federal Food, Drug and Cosmetic Act (as amended 1958, 1960, 1962, 1968, 1976) requires safety and performance testing of all new foods, drugs, food additives and cosmetics. Any substance which shows carcinogenic potential in test animals is barred from the marketplace.

- Federal Insecticide, Fungicide and Rodenticide Act (as amended 1972, 1975, 1978) requires registration of all pesticides, controls their uses and requires certification of applications. Registration of pesticide may be suspended immediately and the product taken off the market if there is an imminent threat to human health or the environment.

- Safe Drinking Water Act (as amended 1977) regulates public drinking water systems and sets standards on finished drinking water to protect the public health from contaminants, including toxic substances. Regulates underground injection of wastes to protect groundwater.

- Marine Protection, Research and Sanctuaries Act (1972) regulates the dumping of any material transported from the United States into ocean waters and any material from outside the U.S. into ocean waters under the jurisdiction of the U.S.

Coordination

The need for coordination of these diverse laws to achieve a unified toxic substances control program is evident. In 1977 the President of the United States instructed the Council on Environmental Quality (the body in the U.S. which has responsibility for developing national environmental policy) to design a coordinated interagency program "(1) to eliminate overlaps and fill gaps in the collection of data on toxic chemicals, and (2) to coordinate federal research and regulatory activities affecting them." In response to this directive, CEQ formed a 17-agency Toxic Substances Strategy Committee of members from federal agencies with major responsibilities for toxic chemicals (10). The committee's work relates to all the laws on toxic chemical controls as shown in Table 1. Examples of issues addressed by the committee include:

- Coordination of regulatory and other approaches to prevent and/or control toxic chemical problems.

- Usefulness of federal research in support of toxics regulation and policy.

- Policies for handling trade secrets and maintaining confidentiality of data.

- Analyses of case histories of past federal actions.

- Coordinated collection and exchange of toxics data among federal agencies.

- Development of a standard procedure or basis for identifying carcinogens throughout the federal government.

- Development of a comprehensive action plan for handling toxic spills and other emergencies, including definitions of state and local government roles.

Closely related to the Strategy Committee is the Interagency Regulatory Liaison Group which was formed by the Environmental Protection Agency, the Food and Drug Administration, the Consumer Product Safety Commission and the Occupational Safety and Health Administration. In 1977 the group published an interagency agreement related to toxic substances control which sets forth a common and consistent approach to:

- Testing protocols, criteria for interpretations, and quality assurance procedures.
- Epidemiological practices and procedures.
- Assessment of risks and estimation of benefits.
- Methods of handling data of mutual interest.
- Research and development policies.
- Regulations and regulatory development activities such as joint public hearings or rulemaking actions.
- Compliance procedures and policies.
- Public information.

Another important coordinating body is the Interagency Toxic Substances Data Committee which is a permanent, independent group co-chaired by CEQ and EPA. This committee has the task of evaluating data needs and coordinating the data systems of all federal agencies generating toxics data into one comprehensive program responsive to nonfederal as well as federal users. This work is being done in support of the Strategy Committee and to fulfill other responsibilities required of EPA and CEQ under the TSCA.

Problems

Based on the variety of programs and breadth of interagency involvement, it is evident that the United States approach to controlling toxic substances is diversified and complex. However, in spite of the many laws, agencies and coordinating efforts there are still many difficult problems to be resolved.

Regulatory actions against toxic and hazardous substances have, for the most part, proceeded on a chemical-by-chemical basis. With the thousands of chemicals involved this is a slow, tedious process which will take years to accomplish. EPA and other agencies are exploring a "generic" approach to toxics regulation. Such an approach would group substances with similar toxic properties—for example, carcinogens or suspected carcinogens—and any current or new substance having these properties would be processed by some common testing and regulatory scheme. A related approach, especially for new chemicals, assumes substances with similar chemical structure have similar effects and control would be focused on groups or classes of chemicals rather than single chemicals. However, the generic regulatory approach is still evolving and to date no single procedural scheme has received strong support.

Most of the data available today has to do with acute effects of specific toxics. There is little knowledge of synergistic or antagonistic effects, environmental transport and fate, biological pathways, and chronic (long-term) effects. A shortage of adequate laboratory facilities and professionals trained in toxicology, industrial hygiene, pathology and related disciplines makes the task of developing adequate toxicological data a very slow, perhaps impossible, process. This problem has yet to be adequately addressed.

The primary purpose of the federal government's toxics program is to bring hazardous wastes of all types under safe management and control. However, a major disposal problem exists, and is getting more critical each day. How do we decide where to destroy, neutralize or contain hazardous wastes? The public demands that these wastes be managed so there is no threat to health and well being, but nobody wants it done in their neighborhood (11). A recent report by the General Accounting Office states that the major roadblock to the building (or even continued operation) of hazardous waste disposal sites is public opposition—not economics, technical feasibility, government bureaucracy, or industrial recalcitrance. Ironically, properly run disposal sites are desperately needed in every state; the lack of adequate sites and guidelines is forcing illegal uncontrolled disposal which has much greater potential for harm to health and the environment.

CONCLUSION

The problems of controlling toxic substances in a democratic, industrialized society are complex, highly diversified and difficult to resolve. However, the problems caused by inadequate control are far greater and the consequences much more severe. The United States has made great strides in developing protective legislation. Now, environmental planning for the decade of the 80s dictates the need for sensitive interaction, communication and cooperation among all levels of government, science and industry. We must develop the data base for making rational decisions and devise reasonable administrative procedures that will protect human health and maintain environmental quality, while preserving economic incentives. And we must have the foresight and confidence to know that the challenge can be met.

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AVAILABILITY AND PLANT UPTAKE OF HEAVY METALS FROM CONTAMINATED DREDGED MATERIAL PLACED IN FLOODED AND UPLAND DISPOSAL ENVIRONMENTS

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ABSTRACT

The availability and plant uptake of heavy metals was evaluated from contaminated dredged material placed in flooded and upland disposal environments using a solid-phase plant bioassay. The objective of the study was to verify previous dredged material research results and to develop a plant bioassay procedure that could indicate phytotoxicity and bioaccumulation of heavy metals in contaminated dredged material. The plant bioassay indicated more uptake and bioaccumulations of cadmium and to a lesser extent zinc when contaminated dredged material was placed in an upland environment where the sediment was allowed to air dry. Placing the contaminated dredged material in a flooded (reduced) environment lowered the availability and plant uptake of cadmium and to a lesser extent zinc. Factors that influenced the availability and plant upake of heavy metals from contaminated sediments included sediment oxidation-reduction potential, organic matter content, total sulfur content, and pH. The plant bioassay showed phytotoxicity and bioaccumulation of arsenic under a flooded environment. Placing the arsenic-contaminated sediment in a upland environment reduced both the phytotoxicity and bioaccumulation of arsenic in the freshwater marsh plant Cyperus esculentus.

INTRODUCTION

Each year the U.S. Army Corps of Engineers (CE) is required to remove in excess of 129 million cubic meters of sediment to maintain navigable waterways in the United States. Increased public concern over the environmental impact of dredging and dredged material disposal led to the establishment of the

Dredged Material Research Program (DMRP) conducted by the U.S. Army Engineer Waterways Experiment Station (WES). The DMRP was initiated in 1973 and was conducted for five years at a cost of $32.8 million. Significant information was obtained on the nature of contaminants in dredged material and the transformations that would occur when dredged material was placed in various disposal environments. The following discussion will attempt to bring the more significant findings together from the DMRP (1) and relate them to the most recent results obtained from ongoing WES research.

Results from laboratory simulations of the effects of redox potential and pH indicated that the availability of heavy metals in sediments was controlled to a large extent by the degree of oxidation of a sediment in combination with the pH of the sediment (2). The metal regulatory processes involved included precipitation with sulfide, adsorption or coprecipitation with colloidal hydrous oxides, and complex formation with soluble and insoluble organics. Cadmium release to soluble and exchangeable forms was shown to be favored by oxidizing conditions, particularly at pH 5.0 and 6.5. As oxidation intensity increased, this cadmium was apparently released from large molecular weight organics.

The Center for Wetland Resources (3) in follow-up research reported that the trace and toxic metal uptake by marsh plants was affected by Eh, pH, and salinity. Plant cadmium content responded more to a change in the physicochemical environment of the rooting medium than did the other metals studied. Plant cadmium content was increased with an increase in oxidation conditions. The highest plant cadmium contents occurred under acid oxidizing conditions. Plant zinc content was also found to increase with increasing oxidation conditions.

While these studies indicate transformations that are most likely to occur under certain redox and pH conditions when heavy metal radioisotopes were added to a sediment, there was a need to develop a technique that could verify the results of laboratory simulations and that could indicate phytotoxicity and bioaccumulation of heavy metals in plants from an inherent contaminated dredged or fill material. This technique could then give CE District personnel a good indication of potential contaminant mobility problems associated with a specific dredged or fill material.

Ongoing WES research is developing techniques to assist CE Districts in estimating the potential bioavailability of contaminants in sediments that require dredging presently or in the near future. One technique being evaluated is essentially a solid-phase plant bioassay to test a sediment for contaminants that are potentially phytotoxic and/or are potentially bioaccumulated in plants. The sediment is tested under both flooded and upland environmental conditions.

METHODOLOGY

The solid-phase plant bioassay was conducted in an experimental unit shown schematically in Figure 1 (4). The sediment to be tested was placed in the inner container. The outer container was used as the water reservoir for the test. Sediments tested under flooded conditions had water of appropriate salinity in the outer container to a height of 5 cm above the sediment surface. Water was added throughout the test period to maintain the sediment under water in a flooded condition. Sediments to be tested under an upland environment were first dried and then placed in the inner container. Deionized water was added to the sediment initially to moisten the sediment and to germinate tubers or to promote seedling growth. Additional water was added only to meet the needs for plant growth. In this way, the sediment remained in an aerobic or upland condition throughout the test period.

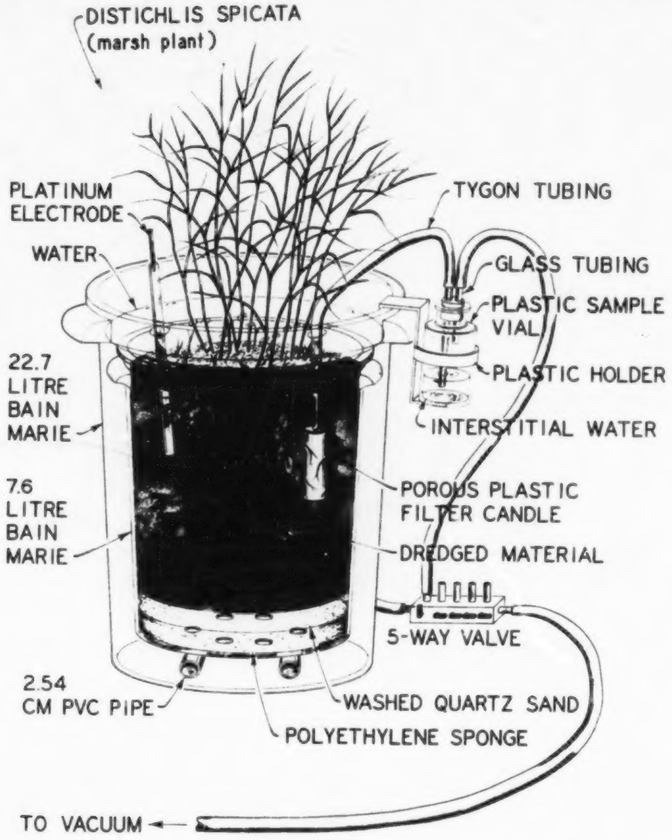

Figure 1. Schematic diagram of the experimental unit used for the solid-phase plant bioassay.

Sediments to be evaluated were collected from five freshwater locations in the Great Lakes and five saltwater locations along the coastline of the United States (Figure 2). These sediments were collected because they were highly contaminated with one or more contaminants. Sediments were analyzed for texture, organic matter, calcium carbonate equivalent, pH, total phosphorus, total nitrogen, oil and grease, sulfur, and heavy metals (total amounts and DTPA extractable).

Plant species used as indicators of phytotoxicity and the bioaccumulation of contaminants were *Cyperus esculentus* in the freshwater sediment tests and *Spartina alterniflora* and *Distichlis spicata* in the saltwater sediment tests. These species have been shown to take up and accumulate heavy metals in previous DMRP research (5). In addition, these species were used in marsh creation projects with dredged material (1). Plants were observed for phytotoxic effects and harvested for yield after the growth period. Plant material was analyzed for heavy metals.

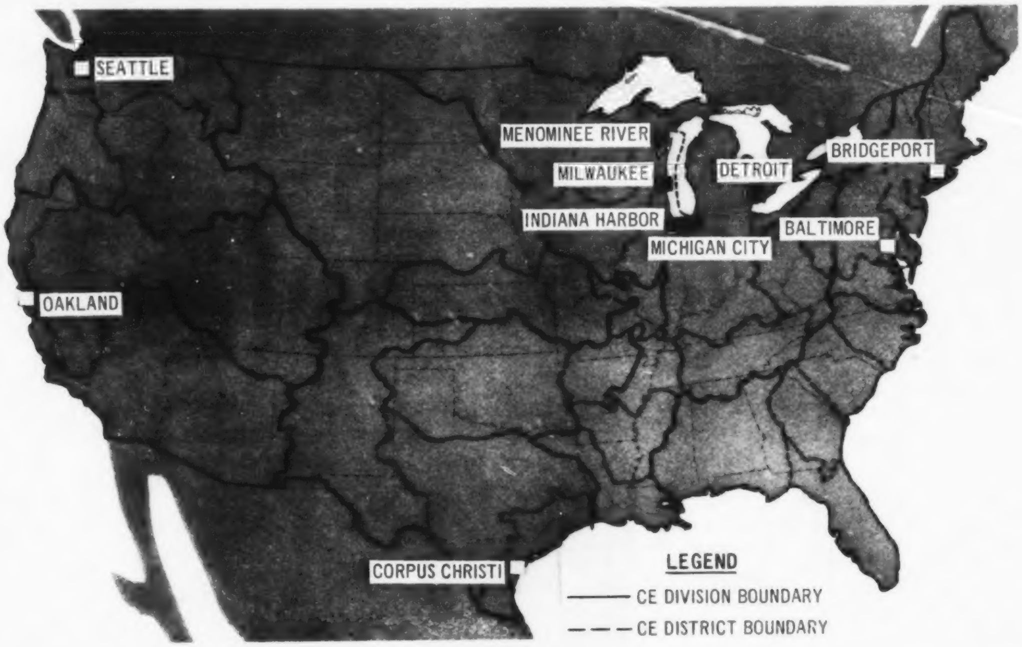

Figure 2. Sediment sample locations for the solid-phase plant bioassay.

RESULTS AND DISCUSSION

Discussion will be limited to the results of the freshwater sediment tests. Occasionally, similar results from saltwater sediment tests will be mentioned.

The texture of both the freshwater and saltwater contaminated sediments was predominantly silt loam (Table 1). This is not surprising since heavy metal contaminants tend to adsorb more readily to finer particles of higher exchange capacity than to coarser sand particles of lesser adsorptive capacity. Most sediments contained organic matter in excess of 5 percent with the exception of DE 3 and ME 1 (Table 1). The saltwater sediments contained a similar range of organic matter. Organic matter has been suggested as having major influence on the availability and translocation of heavy metals through the environment (6,7).

TABLE 1. SELECTED PHYSICAL AND CHEMICAL PARAMETERS OF THE FRESHWATER SEDIMENTS

Location/ Site	Texture	Organic Matter %	CaCO^3 Equivalent %	pH* Flooded	pH* Upland
DE[+] 1	Silt loam	10.7	32.1	7.50	8.50
2	Silt loam	10.7	23.2	7.50	7.59
3	Clay loam	3.1	22.3	7.27	8.07
MC 1	Loam	5.7	19.5	ND†	7.37
2	Silt loam	14.2	13.5	7.23	7.08
3	Loam	13.0	21.1	7.07	7.24
IN 1	Sandy loam	6.6	18.2	7.40	7.25
2	Silt loam	7.3	13.6	7.37	7.58
3	Loam	21.9	16.7	7.50	6.72
MW 1	Silt loam	13.1	34.5	7.40	7.64
2	Silt loam	8.8	51.7	7.20	7.69
3	Silt loam	8.8	56.3	7.40	7.74
ME 1	Sand	2.7	7.6	7.33	7.02
2	Sandy loam	27.3	6.5	7.45	6.38
3	Silt loam	19.5	35.4	7.27	7.02
C.V.‡		5.9	4.5	0.78	0.38

* Flooded pH is pH of the initial interstitial water. Upland pH is pH of a 1:2 sediment-to-solution suspension using air-dried sediment.

[+] DE = Detroit; MC = Michigan City; IN = Indiana Harbor; MW = Milwaukee; ME = Menominee River (see Figure 2).

† ND = not determined.

‡ C.V. = coefficient of variation (%).

All of the freshwater sediments contained large amounts of calcium carbonate and had pH values approximating neutrality (pH 7.0). Under upland (air dried) conditions, the sediments containing the most organic matter showed a reduction in pH from near neutrality to pH 6.38 (Table 1). A similar but more pronounced effect was observed for the saltwater sediments, in that the pH under upland conditions decreased as low as 5.0 in a sediment containing 26.7 percent organic matter, 28.1 mg/g sulfur, and only 0.3 percent calcium carbonate equivalent. In this case there was not sufficient calcium carbonate to buffer the decrease in pH when both the organic matter and sulfur were oxidized under upland conditions. The solubility and availability of some heavy metals such as zinc, copper, iron, manganese, and lead have been shown to decrease with increased soil pH and increased amounts of calcium carbonate (8-13).

The fertility of the sediment can be inferred from the amount of phosphorus (P) and nitrogen (TKN) present (Table 2). All plants grown in the freshwater sediments were nitrogen limited (14). No additional fertilizer was applied to the sediments during the tests. Consequently plants were dependent on whatever was present in the sediment.

Oil and grease ranged from 0.2 to 12.2 mg/g (Table 2). Some of the saltwater sediments tested contained up to 97.5 mg/g of oil and grease. These elevated amounts of oil and grease did not appear to be phytotoxic to plant growth. However, one difficulty was observed when the freshwater sediments were dried and rewet in preparation for planting. The dried sediment repelled

TABLE 2. OTHER SELECTED CHEMICAL PARAMETERS OF THE FRESHWATER SEDIMENTS

Location/ Site	Concentration (mg/g)			
	Total P	TKN	Oil and Grease	Total Sulfur
DE[+] 1	1.11	0.68	7.8	1.7
2	2.38	2.32	9.2	1.9
3	0.44	0.68	0.2	3.5
MC 1	0.70	1.01	0.5	1.6
2	2.56	3.77	5.8	4.1
3	2.11	3.74	4.6	3.8
IN 1	0.74	0.94	4.4	1.6
2	0.73	0.63	12.2	6.0
3	5.60	3.05	6.8	7.2
MW 1	1.55	2.93	4.8	2.1
2	0.49	3.07	0.2	3.7
3	0.55	6.33	0.2	1.3
ME 1	0.22	0.47	0.22	0.3
2	0.39	2.08	1.5	2.0
3	1.34	2.79	2.2	2.3
C.V.*	10.5	4.8	54	27

[+] DE = Detroit; MC = Michigan City; IN = Indiana Harbor; MW = Milwaukee; ME = Menominee River (see Figure 2).

* C.V. = Coefficient of variation (%).

the applied water and was extremely difficult to rewet. The presence of the oil and grease appeared to inhibit the rewetting process.

The sulfur content of the freshwater sediments ranged from 0.3 to 7.2 mg/g (Table 2). The saltwater sediments contained much more sulfur, ranging from 2.3 to 28.1 mg/g. Upon air drying, the pH of the saltwater sediments decreased below neutrality whenever the sulfur content was greater than 13 mg/g and the calcium carbonate equivalent was less than 0.4 percent. Saltwater sediments may be prone to show reduction in pH upon drying if placed in an upland environment.

The redox potential of one of the freshwater sediments under flooded and upland conditions during the tests is shown in Figure 3. Similar conditions were observed for the other freshwater sediments. Flooded conditions resulted in a stable negative redox potential throughout the study. The redox potential under upland conditions oscillated up and down due to the application of water to meet the needs for plant growth. However, for the majority of the growth period the redox potentials of the upland sediments were always higher than those of the flooded sediments.

The concentrations of heavy metals found in the freshwater sediments are shown in Table 3. The saltwater sediments contained similar ranges. Total nitric acid digests of the sediments indicated how much heavy metals were present but did not indicate the bioavailability of the heavy metals.

In order to get an indication of the bioavailability of sediment heavy metals under both flooded and upland conditions, a DTPA extraction procedure was used. The DTPA extraction described by Lee et al. (15) has been shown to estimate plant-available zinc, copper, manganese, and iron for agronomic plants (16) and to correlate well to plant content of zinc, cadmium, and copper and to a lesser extent with lead and chromium in saltwater marsh plants (15).

While the DTPA extraction data for copper, mercury, nickel, chromium, and lead did not show consistent differences, DTPA-extractable zinc, cadmium, manganese, iron, and arsenic showed consistent results. Table 4 shows the concentrations of zinc, cadmium, and arsenic in the DTPA extracts of the freshwater sediments. Concentrations of zinc and cadmium in DTPA extracts of saltwater sediments are shown in Table 5. While DTPA-extractable zinc appeared to be higher under upland conditions in 10 out of 15 freshwater sediments (Table 4), all of the saltwater sediments showed pronounced increases in DTPA-extractable zinc under upland conditions (Table 5). A similar observation can be made for DTPA-extractable cadmium.

More cadmium was extracted under upland conditions than under flooded conditions, expecially in the saltwater sediments (Table 5). These results verify previous DMRP research that showed placing dredged sediment under oxidized environments potentially increased the amount of available zinc and cadmium in the sediment.

The opposite effect was observed with regard to DTPA-extractable arsenic (Table 4). More arsenic was extracted under flooded conditions than upland conditions.

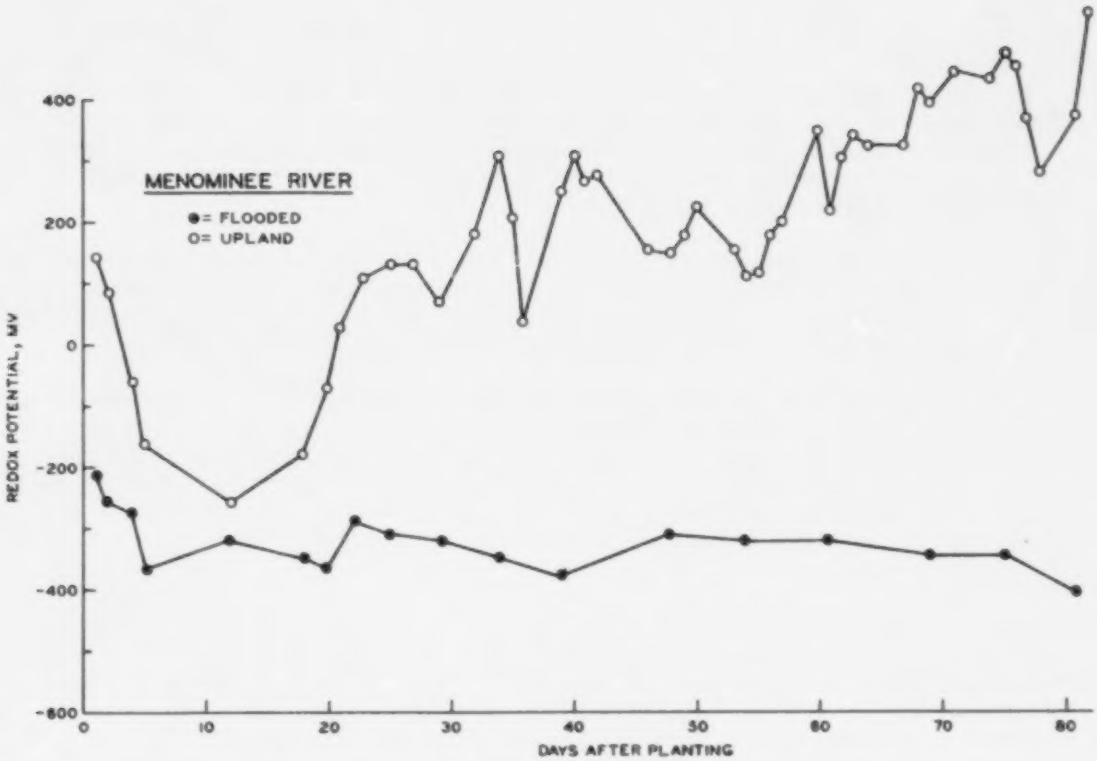

Figure 3. Redox potential during growth period of Cyperus esculentus in flooded and upland disposal environments.

While the DTPA showed important results, plant content of zinc, cadmium, and arsenic were even more important (Table 6). While certain freshwater sediments such as MC and IN showed higher plant leaf zinc content under upland conditions, all of the sediments showed higher plant leaf cadmium content

TABLE 3. TOTAL* CONCENTRATIONS OF HEAVY METALS IN FRESHWATER SEDIMENTS

Location/ Site	Concentration(μg/g)									
	Zinc	Cadmium	Copper	Nickel	Chromium	Lead	Mercury	Iron†	Manganese	Arsenic
DE+ 1	2423	29.5	199	62.5	94.8	226.1	0.36	74.31	1203	11.4
2	2048	21.5	202	152.8	327.5	326.0	0.56	56.97	536	7.5
3	233	7.7	22	--	--	--	0.13	17.10	401	3.3
MC 1	796	6.2	21	20.0	9.5	30.0	0.06	11.56	258	2.5
2	1857	39.8	115	--	--	--	1.02	19.73	518	4.7
3	978	35.9	121	105.3	135.5	656.5	0.18	29.81	523	2.3
IN 1	660	7.6	35	22.0	96.3	99.4	0.08	40.17	638	18.8
2	1433	16.0	65	37.3	117.3	214.7	0.29	89.81	1461	27.0
3	8867	45.6	340	107.2	1974.3	1521.0	1.50	291.31	2503	37.5
MW 1	921	14.8	82	48.7	231.3	385.0	0.60	23.91	541	7.9
2	31458	7.8	16	19.4	23.5	25.5	0.01	12.67	523	1.1
3	711	5.8	12	14.2	10.0	26.8	0.00	9.86	469	0.0
ME 1	124	0.1	10	8.2	10.5	143.0	0.05	7.02	164	4.1
2	3992	0.1	24	10.5	8.2	315.3	0.27	8.56	267	6.4
3	230	9.3	37	19.9	23.7	60.0	0.44	27.41	1213	316.5
C.V. ‡	61	25	17	11	53	40	77	6.9	9.8	26

* Nitric acid digest for total heavy metal content (4).

+ DE = Detroit; MC = Michigan City; IN = Indiana Harbor; MW = Milwaukee; ME = Menominee River (see Figure 2).

† Concentration of iron is in mg/g

‡ C.V. = coefficient of variation (%).

TABLE 4. DTPA EXTRACTABLE HEAVY METALS FROM FLOODED AND UPLAND FRESHWATER SEDIMENTS

Location/ Site	Concentration (μg/g)					
	Zinc		Cadmium		Arsenic	
	Flooded	Upland	Flooded	Upland	Flooded	Upland
DE[+] 1	48.1	173.1	<0.0005	6.30	<0.005	CI[*]
2	276.6	255.1	4.89	4.15	0.169	CI
3	1.8	13.6	0.30	0.03	<0.005	CI
MC 1	179.9	130.0	3.33	2.92	<0.005	CI
2	180.1	811.6	0.83	25.26	<0.005	CI
3	5.4	743.1	<0.0005	31.25	<0.005	CI
IN 1	205.8	227.0	0.53	1.10	<0.005	CI
2	48.3	512.9	<0.0005	3.11	<0.005	CI
3	10.6	1339.2	<0.0005	5.44	<0.005	CI
MW 1	144.6	265.6	3.39	6.08	0.122	CI
2	21.1	14.9	0.82	0.73	<0.005	CI
3	14.5	14.4	0.50	0.53	<0.005	CI
ME 1	5.3	6.1	0.40	0.19	<0.005	CI
2	5.3	27.1	<0.0005	0.64	0.278	CI
3	6.3	33.5	<0.0005	2.72	131.000	37.200
C.V.[†]	10.4	14.2	34.1	18.1	56.7	6.2

[*] CI = Chemical interference

[+] DE = Detroit; MC = Michigan City; IN = Indiana Harbor; MW = Milwaukee; ME = Menominee River (see Figure 2).

[†] C.V. = Coefficient of variation (%).

under upland conditions than under flooded conditions. Even though plant leaf cadmium content reached 20.84 μg/g in the MC 2 sediment, no phytotoxic symptoms were observed.

These results indicate that sediments containing elevated amounts of zinc and cadmium would have the availability of zinc and cadmium lowered if these sediments were placed in a flooded (reduced) environment. Placing these sediments in an upland environment would result in these metals becoming more available for plant uptake as well as for migration into the surrounding environment.

Phytotoxic symptoms were observed (Figure 4) in freshwater plants containing 10.7 μg/g arsenic (Table 6). Plant growth was better when the same

TABLE 5. DTPA EXTRACTABLE HEAVY METALS FROM FLOODED AND UPLAND SALTWATER SEDIMENTS

Location/ Site	Concentration (μg/g)			
	Zinc		Cadmium	
	Flooded	Upland	Flooded	Upland
BR[+] 1	0.3	408.3	<0.0005	0.42
2	0.9	803.2	<0.0005	28.54
BA 1	0.3	455.3	<0.0005	32.34
2	0.1	954.0	<0.0005	15.94
3	0.6	134.2	<0.0005	2.06
CC 1	10.8	93.9	<0.0005	1.16
2	54.1	305.5	0.22	3.27
3	5.6	1511.8	<0.0005	19.02
OH 1	4.5	125.4	0.86	2.08
2	1.0	166.9	<0.0005	1.96
3	70.07	168.4	0.38	0.75
SE 1	5.6	67.5	0.18	1.01
2	3.1	128.5	<0.0005	2.30
3	0.1	111.6	<0.0005	2.03
C.V.*	40.3	22.0	75.8	42.4

[+] BR = Bridgeport; BA = Baltimore; CC = Corpus Christi; OH = Oakland; SE = Seattle (see Figure 2).

* C.V. = Coefficient of variation (%).

sediment was placed in the upland environment (Figure 4). Plant leaf arsenic content was reduced to 1.45 μg/g under upland conditions. Availability of arsenic was similar to that of phosphorus in that arsenic was more available under flooded (reduced) conditions and became precipitated and adsorbed to soil particles as the sediment or soil dried out (17). Sediments containing elevated amounts of arsenic would have the availability of arsenic reduced if placed in an upland environment.

Sediment parameters that appeared to be influencing the availability of heavy metals in contaminated sediments were redox potential, organic matter content, total sulfur content, manganese and iron content, calcium carbonate content, and pH. Extraction of heavy metals from sediments with DTPA was also directly related to plant content of heavy metals.

The results of the solid-phase plant bioassay verified the findings from the DMRP. The test is easy to conduct and gives a good indication of phyto-

TABLE 6. HEAVY METAL CONTENT OF CYPERUS ESCULENTUS LEAVES AFTER PLANT GROWTH IN FRESHWATER SEDIMENTS UNDER FLOODED AND UPLAND ENVIRONMENTS

Location/ Site	Concentration (µg/g)					
	Zinc		Cadmium		Arsenic	
	Flooded	Upland	Flooded	Upland	Flooded	Upland
DE[+] 1	75.5	75.8	0.77	2.81	<0.025	<0.025
2	96.8	122.8	0.79	6.49	<0.025	<0.025
3	151.5	150.8	0.23	1.17	<0.025	<0.025
MC 1	63.3	259.6	1.31	17.64	<0.025	<0.025
2	70.04	168.3	2.81	20.84	<0.025	<0.025
3	56.7	84.5	2.50	7.80	<0.025	<0.025
IN 1	34.8	112.8	0.19	1.95	<0.025	<0.025
2	51.8	154.0	0.51	6.34	<0.025	<0.025
3	63.5	172.9	0.25	1.27	<0.025	<0.025
MW 1	48.8	256.4	1.24	12.60	<0.025	<0.025
2	87.8	83.3	0.41	1.84	<0.025	<0.025
3	84.9	58.0	0.32	0.87	<0.025	<0.025
ME 1	29.6	23.8	0.82	1.45	<0.025	<0.025
2	76.5	80.8	0.36	1.44	<0.025	<0.025
3	0.2	17.3	0.47	9.57	10.700	1.450
C.V.*	26.6	24.5	40.2	26.8	50.2	88.4

[+] DE = Detroit; MC = Michigan City; IN = Indiana Harbor; MW = Milwaukee; ME = Menominee River (see Figure 2).

* C.V. = coefficient of variation (%).

toxicity and bioaccumulation of contaminants in sediments placed in flooded and upland disposal environments. The plant bioassay procedure is presently being refined and will be verified with field tests in the future.

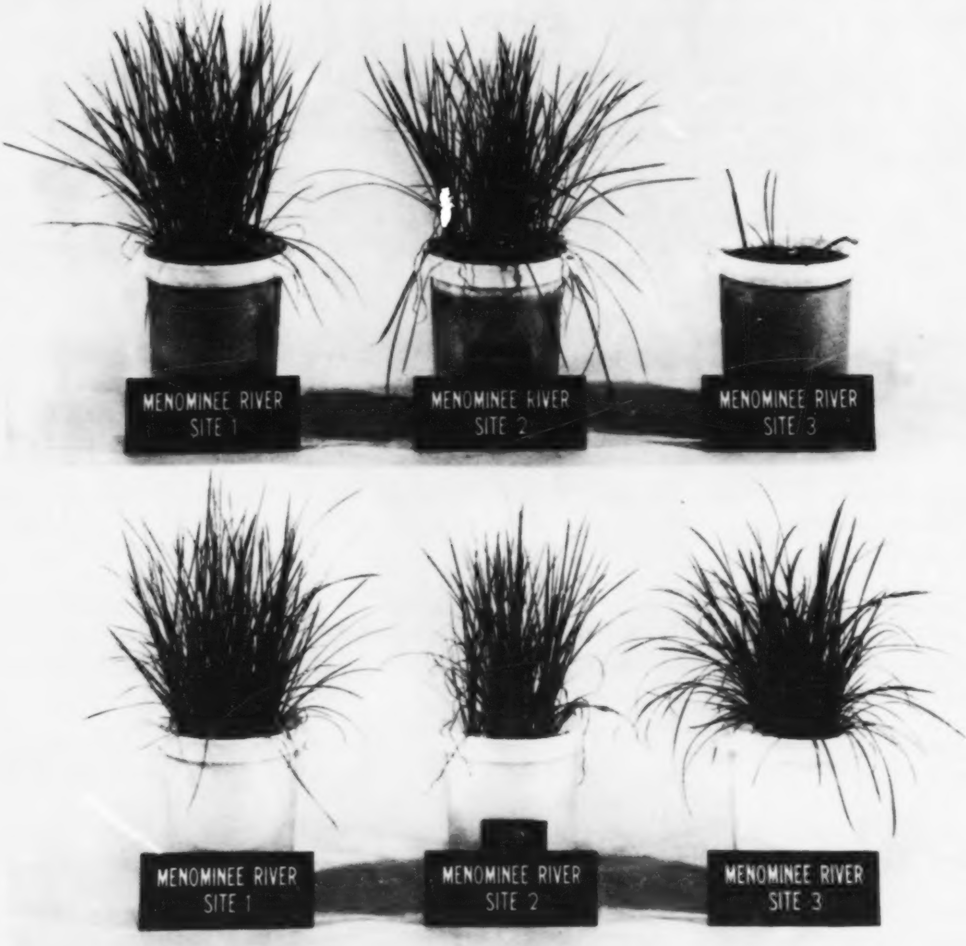

Figure 4. Growth of Cyperus esculentus under flooded and upland disposal environments in sediments from three sites on the Menominee River.

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DISTRIBUTION AND CONCENTRATION OF PCB IN THE HUDSON RIVER AND ASSOCIATED MANAGEMENT PROBLEMS

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ABSTRACT

The Hudson River is contaminated with PCBs and a remedial program has been formulated by New York State to cope with this toxic pollutant. Over $3 million has been spent to fully document the extent of the PCB contamination in the Upper Hudson River. It appears that over 60 percent of the PCBs in the riverbed are contained within 40 so-called "hot spot" areas, that is, those areas that have PCB concentrations greater than 50 ppm. After evaluating all alternatives the PCB Reclamation Project was formulated which consists of mechanically or hydraulically dredging the 40 "hot spot" areas and placing the contaminated sediments in an uncapsulated land burial facility. Such a facility would meet all Federal and State criteria for PCBs disposal. New York is seeking to finance this remedial project through the use of Federal funds, that could be obtained either through the Clean Water Act or through special Congressional legislation or through the Superfund Bill, once it is enacted. The PCB Reclamation Project, if funded this spring as is presently anticipated, could be completed by 1982.

INTRODUCTION

The Hudson River is contaminated with polychlorinated biphenyls (PCBs). The toxicity of PCBs has been a topic of much research and experiments have shown that PCBs at high doses can cause death and liver tumors have also been induced in mice and rats. Much of the information on the toxicity of PCBs can be found in the Criteria Document for PCBs (1976) published by the Environmental Protection Agency (1) and a report published by the United States Department of Health, Education and Welfare (2).

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It is currently estimated that 640,000 pounds of PCBs are contaminating the sediments of the Hudson River, from New York City to the Fort Edward Dam site at Fort Edward, New York. PCB contamination of the Hudson River poses a potential threat to the health of thousands of New Yorkers, especially those who obtain their drinking water directly from the river.

Also, the United States Food and Drug Administration has set allowable limits for PCBs in food, including fish. As a result of the high PCB concentration in fish, the New York State Department of Environmental Conservation has banned commercial fishing for certain species in the lower Hudson, from Troy to New York City and both commercial and recreational fishing for all species from Troy to Fort Edward.

If remedial action is not taken, it is likely that the Hudson River will remain contaminated with PCBs for a century or more.

BACKGROUND

Polychlorinated biphenyls (PCBs) were first manufactured in 1929 and were soon found to be ideal for a number of industrial uses. They are very stable chemically and biologically, have a low electrical conductivity and are nearly insoluble in water. Because of these properties, PCB usage by General Electric had been extensive and long standing at their capacitor manufacturing facilities at Fort Edward and Hudson Falls.

Over 78 million pounds (3) of PCBs were purchased by these two facilities between 1966 and 1974. Although records do not exist for years prior to 1966, PCBs were used at the Fort Edward and Hudson Falls plants for more than 25 years.

High levels of PCBs in the Hudson River biota were first reported in 1969, but the seriousness of the situation was not recognized for several years. Extensive sampling of the river in 1975 implicated the General Electric plants as the major source of PCBs in the Upper Hudson River.

Acting on information supplied by the Environmental Protection Agency and the Fish and Wildlife Service, and on additional evidence collected by the Department of Environmental Conservation, the General Electric Company (GE) was charged with polluting the river with the toxic substance known as PCB. Administrative proceedings between the Environmental Conservation Department and GE began on September 8, 1975.

On February 9, 1976, after weeks of testimony, reports, studies and other exhibits, Professor Abraham D. Sofaer (4), the Hearing Officer, found GE was responsible for the high concentrations of PCBs in the Upper Hudson's waters, sediments, organisms and fish. A settlement agreeable to all parties was negotiated and signed on September 8, 1976.

Under the terms of the settlement, a comprehensive program of at least $7 million was to be enacted in order to deal with PCBs in the Hudson River and with related environmental concerns. GE was to gradually eliminate all PCB discharges from the two plants by July 1977. In addition, GE agreed to contribute $3 million to the Environmental Conservation Department as its share of a program to monitor the presence and levels of PCBs in the Hudson; to further investigate the need for remedial action concerning PCBs in the river; and aid in developing a program to regulate the storage and discharge of environmentally hazardous substances.

New York State agreed under the terms of the settlement, to contribute an additional $3 million for the above mentioned work. Also, GE was required to perform $1 million of research related to PCBs, including a study of the environmental compatability of a PCB substitute.

PROBLEM

Studies (Table 1) directed by NYS Environmental Conservation Department have reinforced the early evidence of PCB contamination of the Hudson River. Of the more than 600,000 pounds of PCBs existing in the river, approximately two thirds of this amount are still located in the bed sediments north of Troy. It has also been calculated that more than 5,000 pounds of PCBs move over the Federal Dam at Troy, New York and into the estuarine portion of the river.

Analysis of edible portions of fish has exceeded many times the FDA tolerance level of 5 ppm. Water column concentrations of PCBs measure around 1 ppb and organisms have been shown to accumulate PCBs rapidly from the river water. Volatilization of the various PCBs aroclors occurs more readily than was originally predicted and air pollution from PCBs was found to be a reality. The settlement funded studies paint a bleak picture of the total environment for the Hudson River Basin (Table 2).

Large amounts of PCBs were caught in the sediments built up behind dams near the capacitor plants at Fort Edward and Hudson Falls (Figure 1). The first dam, located at Fort Edward, was removed in 1973, allowing large amounts of contaminated sediments to move downstream.

The scientific and engineering studies have pinpointed 40 riverbed sediment areas containing more than 50 parts per million of PCBs. These 40 areas, known as the "hot spots", (Figure 2) and the remnant deposits, constitute only 8 percent of the total upper riverbed, but contain approximately 60 percent of all of the PCBs located in the Upper Hudson Riverbed sediments.

The remaining dam sediments behind the Fort Edward Dam, containing an additional 28 percent of the upper river's PCBs, now form part of the riverbank south of Fort Edward.

A summary of distribution of residual PCBs in the Hudson River Basin is provided by Table 3.

TABLE 1. HUDSON RIVER PCB SETTLEMENT STUDIES (4, 5) (1976 to present)

- I. Aquatic Studies
  - A. Physical
    - 1. Monitoring of river flow and sediment and PCB transport - USGS
    - 2. PCB mapping, upper river - Normandeau Associates
    - 3. Bedload sediment transport - Rensselaer Polytechnic Institute
    - 4. Screening survey of lower river PCB concentrations - EPA
    - 5. PCB concentrations of estuary sediments - Lamont Doherty
    - 6. Groundwater - Weston
    - 7. Wastewater - Pure Waters, O'Brien and Gere
    - 8. Use of high volume centrifuge to better define PCB-particulate/water interchange - DEC Bureau of Water Research
    - 9. Additional bed sediment sampling - DEC Bureau of Water Research
  - B. Biological
    - 1. Fish monitoring - fish collections and data evaluation - Dec, PCB analysis by O'Brien and Gere
    - 2. Macroinvertebrates monitoring - DOH
    - 3. Aquatic food chain dynamics and lower trophic level studies - NYU Medical Center, SUNY Stony Brook and Fordham University
- II. Land
  - A. Physical
    - 1. Air monitoring - DEC Divsion of Air Resources
  - B. Biological
    - 1. Plant and Farm Product uptake - Sample collection and data evaluation by DEC Bureau of Water Research and Boyce Thompson Institute, PCB analysis by Raltech
- III. Engineering related to remedial measures
  - A. Hot spot dredging project - Malcolm Pirnie, Inc.
  - B. Landfills and Dumps - Weston
  - C. Alternatives: "No action", Lawler, Matusky and Skelly
    Effects of remedial action - Hydroscience
  - D. Removal and Treatment - GE
  - E. Public Water Supply - Treatability Study - NYS Dept of Health and O'Brien and Gere
- IV. Project Management
  - A. Study Management and data storage - DEC Bureau of Water Research
  - B. Laboratory intercomparison and quality control - NYS Department Health - Division of Laboratories and Research
  - C. Modeling
    - 1. Up river sediment transport modeling - Lawler, Matusky and Skelly
    - 2. Biological modeling - Hydroscience
  - D. Study Interpretation and Report Preparation - DEC Bureau of Water Research

TABLE 2. HUDSON RIVER SYSTEM TYPICAL PCB VALUES

Upper Hudson River		
Bed Sediment	20-150	PPM
Water Column	0.1-1	PPB
Fish	10-130	PPM
Macroinvertebrates	3-10	PPM
Dredge Spoil Areas		
Spoil	5-50	PPM
Leachate	0.5	PPB
GE Industrial Landfills		
Waste Material	500-5000	PPM
Leachate	50-500	PPB
Ambient Air (Fort Edward)	100-3000	Nanograms/m^3
Dust (Fort Edward)	17	PPM
Plants		
Near GE Landfill	10-500	PPM
Near Dredge Spoil Area	0.2-1.3	PPM
Lower Hudson River		
Bed Sediment	1-15	PPM
Water Column	ND-0.8	PPB
Fish		
Resident	5-10	PPM
Migratory	0.5-15	PPM
Macroinvertebrates	1-3	PPM
Turtles		
Muscle	5	PPM
Eggs	25	PPM

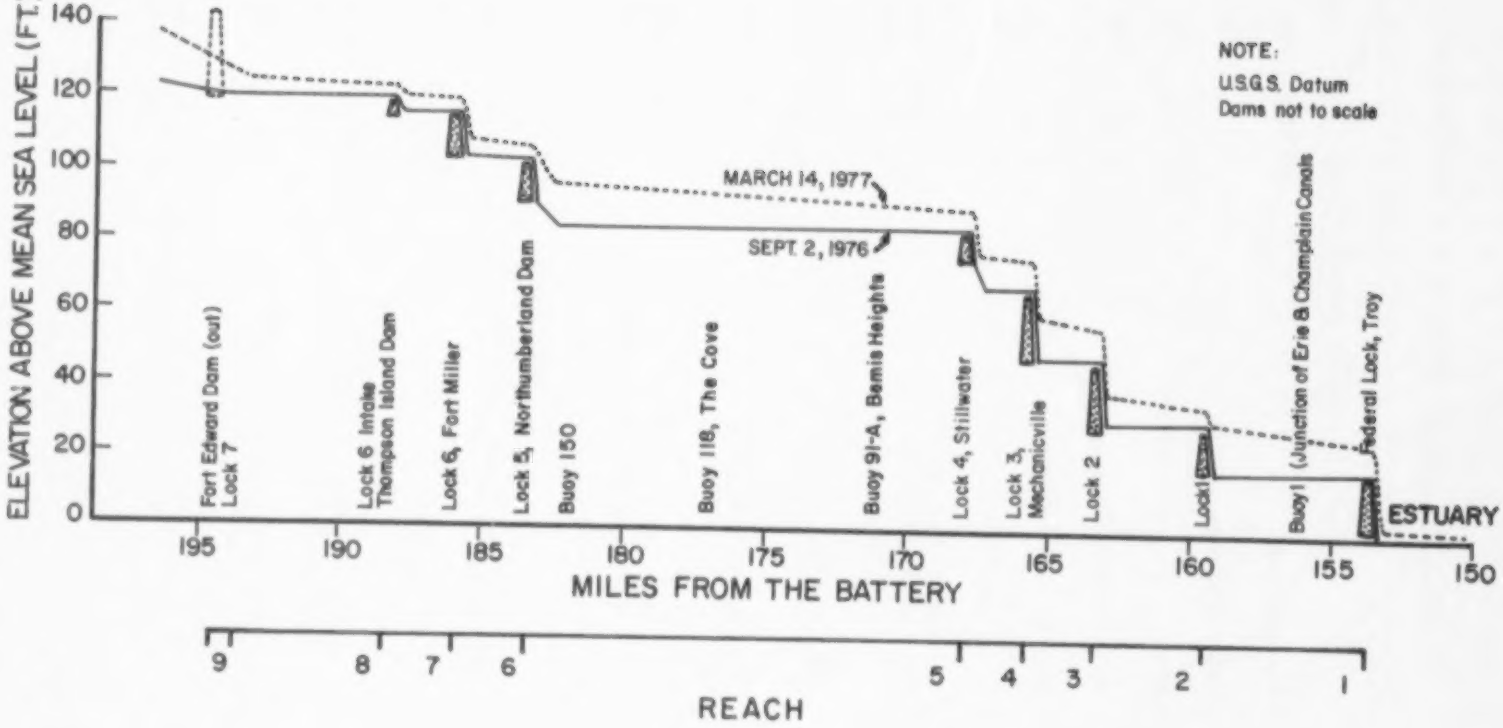

Figure 1. Upper Hudson River water surface profile (Fort Edward to Federal Lock).

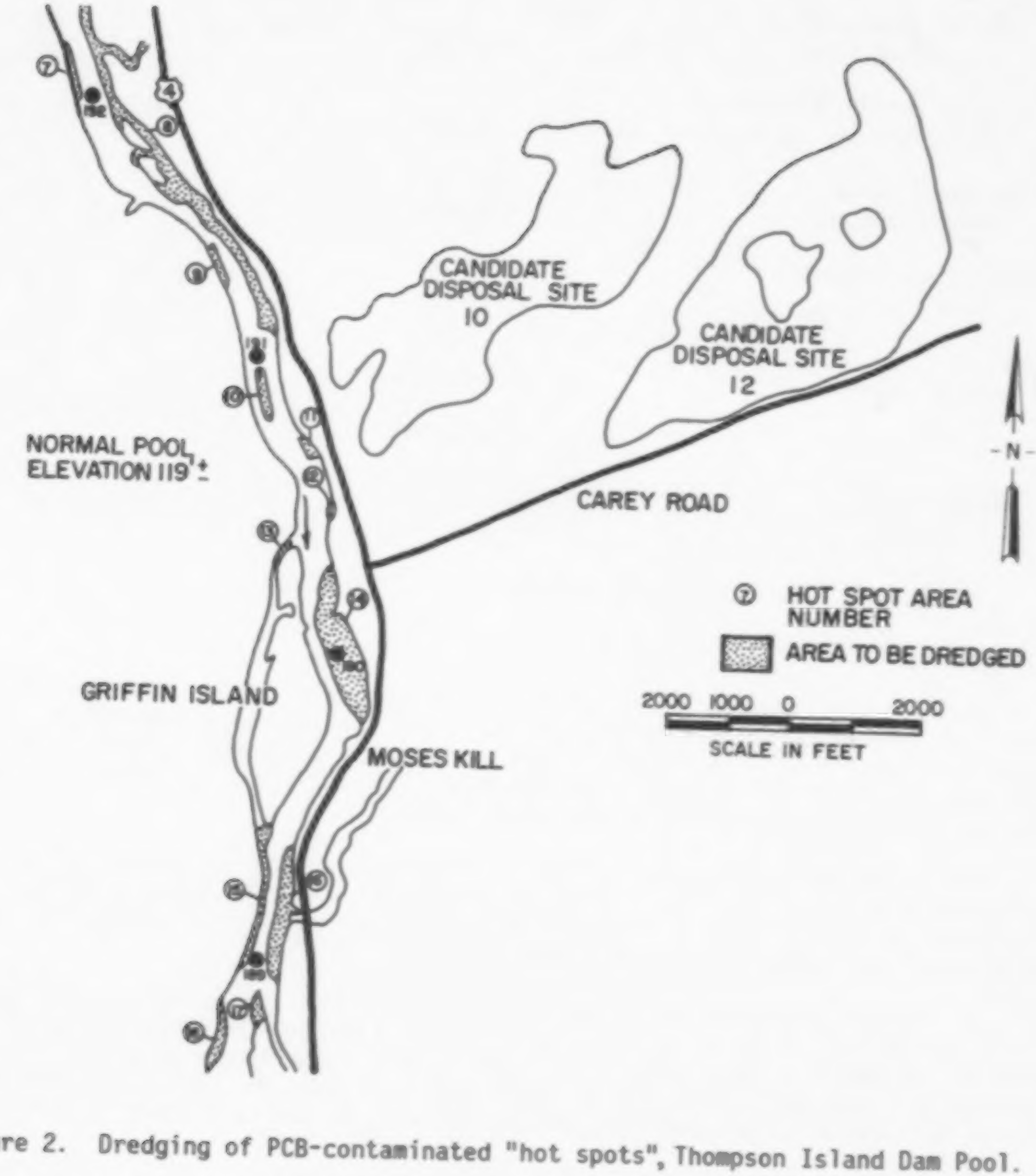

Figure 2. Dredging of PCB-contaminated "hot spots", Thompson Island Dam Pool.

TABLE 3. SUMMARY DISTRIBUTION OF RESIDUAL PCBs IN THE HUDSON RIVER BASIN (5)

Area	Estimated total PCBs lbs	Estimated Scourable PCBs lbs	Calculated loss rate* (lbs/yr)
River sediments			
Remnant deposits	140,000	45,000	8,600
Upper Hudson Riverbed (Ft. Edward Dam Site to Troy)	300,000	98,000	5,700
Lower Hudson Riverbed (Estuary-Troy to New York Harbor)	200,000	?	?
Dredge spoil areas (Upper Hudson)	160,000	-	170
Landfills and dumps (Upper Hudson)	530,000	-	800
Biota			
Lower River	200-2,000	-	0
Total	1,330,000		

*Does not include volatilization. (Based on April 1978 Data).

DISTRIBUTION AND CONCENTRATIONS OF PCB

Methods

Between 1974-1979, a large number of bed sediment samples have been taken from the Hudson River and analyzed for PCBs. Normandeau Associates, Inc. (NAI) did the primary bed sampling and mapping in the summer of 1977, taking 670 grab and 200 core samples. In addition, DEC staff has collected over 200 grab samples of bed sediments in the Upper Hudson.

NAI collected their grab samples on transects across the river, while the cores were taken in the finer textured sediments near the banks. Because of the known sources of PCB contamination, sampling was more frequent near Fort Edward and less frequent down the river.

NAI used a Motorola miniranger and transponder system to provide quick and accurate sample locations and a precision Rayethon DE-719B fathometer to provide accurate bottom transects. The pulsed radar signals are timed and interpolated automatically and printed out on a tape in the boat. For sampling, NAI used a shipek grab sampler and manually forced core tubes.

DEC personnel used a Leitz range finder with accuracy comparable to a transit and stadia system for horizontal locations. DEC also used a ponar grab sampler, manually forced core tubes and a 7 m long copper probing pipe 1.5 cm in diameter. The probing pipe proved to be a very quick and reliable tool in determining the depth and nature of the sediments. Cores were typically cut into 5-15 cm (2-6 in.) sections. All sediment samples were given a visual texture code on a scale ranging from 0 for clay to 9 for coarse sand.

Sediment analysis included PCB, total solids, volatile solids and texture. PCB was reported to a 1 μg/g detection limit as three aroclors - 1016, 1221 and 1254 and their total, and a rigorous quality control program was maintained at all times (6). It appeared that the shaker extraction used for PCB gave about 85 percent recovery compared to the soxhlet method. The data including exact locations, depths, and texture were entered on computers (7, 8).

Results

PCB concentration in the Upper Hudson River sediments appear to be log normally distributed (Figure 3). Thus, for statistical analysis, log PCB values were used, while for total mass tabulations, a duel area averaging system for hot areas greater than 50 μg/g and cold areas less than 50 μg/g was used. Analysis of the data for each of the two groups, that is, hot areas and cold areas, tends to be more normally distributed than the entire data group.

The areas and arithmetic mean of PCB concentration for each of the hot and cold spots are given in Table 4.

TABLE 4. THE AREA AND AVERAGE PCB CONCENTRATION BY REACH OF HOT SPOTS (concentration > 50 ppm) AND COLD AREAS (concentration < 50 ppm) IN SEDIMENTS OF THE UPPER HUDSON RIVER

		Reaches*							
	Total	8	7	6	5	4	3	2	1
PCB Levels**									
Hot spots	--	151	103	163	70	108	159	--	--
Cold areas	--	19	20	20	13	13	23	13.9	9.7
Overall	--	67.6	86.2	43.6	11.6	41.9	52.3	13.9	9.7
Area ($x10^6 ft^2$)									
Hot spots	10.86	4.27	0.6	2.59	1.75	1.5	0.15	0	0
Cold areas	120.79	10.73	7.4	5.91	41.2	9.7	10.85	15	20
Total	131.65	15.00	8.0	8.5	42.95	11.2	11.0	15	20

* Reaches are identified in Figure 1

** Average concentrations for sediment grab samples in ppm.

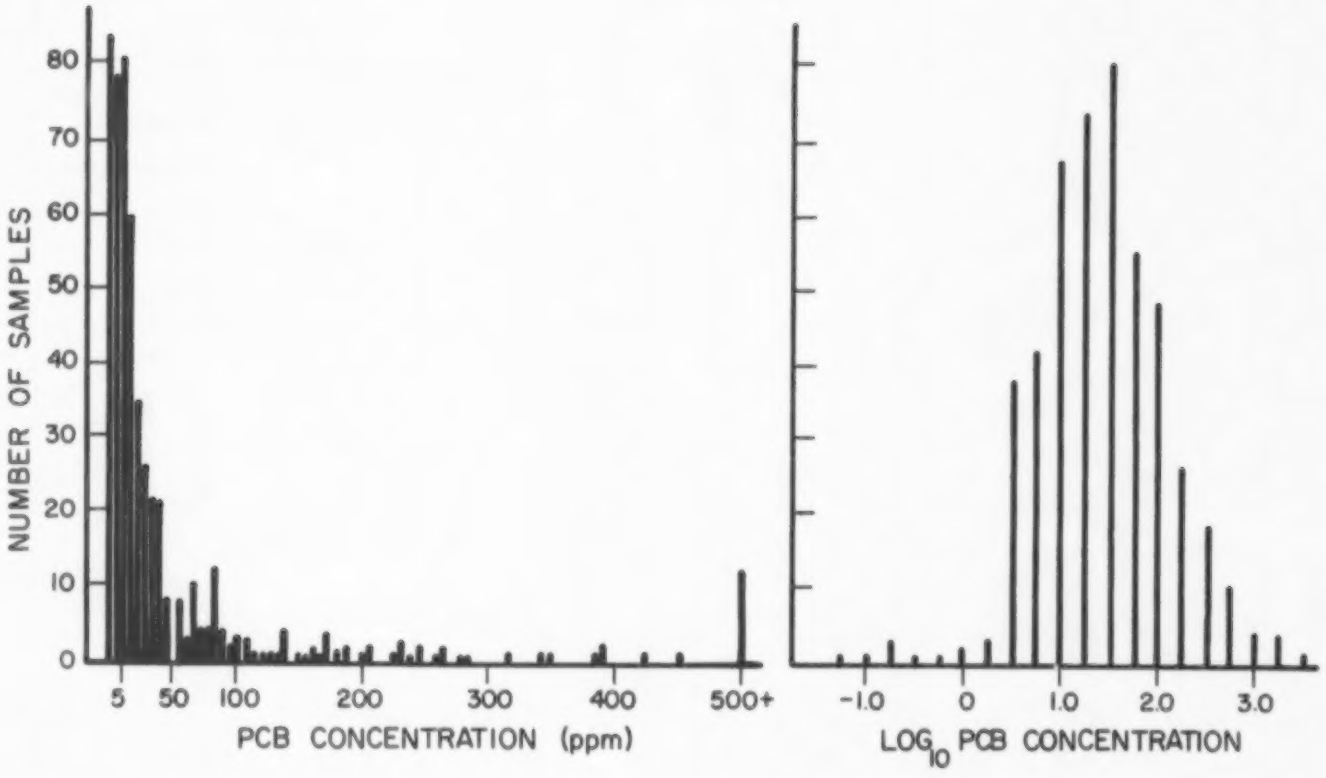

Figure 3. Frequency distribution of sediment PCB levels in the Upper Hudson River.

PCB in µg/g was typically higher in the finer textured sediments near the banks and lower in the coarse sediments in the main channel. The downriver variation of PCB values was much less than the cross river variation as noted in Figure 4 which shows the log mean PCB values for the grab samples by river reaches. Much of the variation indicated by the 95 percent confidence interval lines are due to across river variations.

It should be noted that Reaches 8, 7, 6 were significantly higher in PCB than Reaches 5, 2, 1 and 0. The drop in PCB in Reach 5 appeared partially due to the nature of the reach which was typically narrower and higher in velocity. Reaches 3 and 4 were wider and lower in velocity than Reach 5.

Additional analysis of the river sediment samples indicated that muck and wood chips had PCB levels typically greater than 50 ppm and very seldom less than 25 ppm. On the other hand, samples consisting of primarily gravel were low in PCB concentrations but would immediately increase if wood chips were present in the gravel.

For the lower river reaches, the sediment did not generally contain wood chips. Three size fractions for 11 samples were compared for PCB and volatile solids. The coarsest size fraction contained a significantly higher percent of volatile solids due to wood chips and also the highest average PCB content. PCB was typically skip graded in sieved samples, that is, concentrations were high in the coarse sand size, low in the fine sand size, and high again in the silt size. The coarse fractions with over 90 percent volatile solids were mostly wood chips. Sieve analysis and texture codes on about 670 NAI grabs were entered on the consultants' computers. The mean D_{50} for these grabs was .3 mm (7). The D_{50} is the sieve size that passes 50 percent of the sample. In addition, 20 percent of the grabs had a D_{50} of greater than 2 mm, while 10 percent of the grabs had a D_{50} of less than .06 mm (the silt size division). The adsorption ability for PCB of various materials, such as activated sludge, Hudson River sediments, top soil, silt, saw dust, celite, etc. was compared, and it was noted that silty soil, wood chips and river sediment adsorbed well compared to such materials as Ottawa sand.

While the grabs were used to assess the spread of PCB across river areas, the cores were used to assess the depth of penetration of PCB. Most core samples were taken in silt and sand near the banks. However, probings were made in both center of channel and near bank regions to determine the depths of sediment present over rock and clay. The PCB typically peaked between an 8 and 30 cm depth in the cores taken in the hot spots or near bank contaminated areas (Figure 5). In the core sampling it was noted that compression of the sediment in the core tube commonly occurred. It is possible that the actual depth of PCB penetration is up to 30 percent greater than the stated depth. The results on the depth of PCB in the main channel were not yet available but probing data indicated that there was on the average less than .52 m (1.7 ft.) of sediment present and it was coarser textured. From limited data, the mass of PCB in cold areas was estimated to be located at the .3 m (1.0 ft.) depth.

One set of 12 cores, the NAI winter cores, were sectioned every inch and selected ones analyzed for PCB, Cs 137, heavy metals and size analysis. The Cs 137 came from fallout from atomic bomb testing and can be used for roughly dating the sediment layers. It appeared that these near-bank areas received

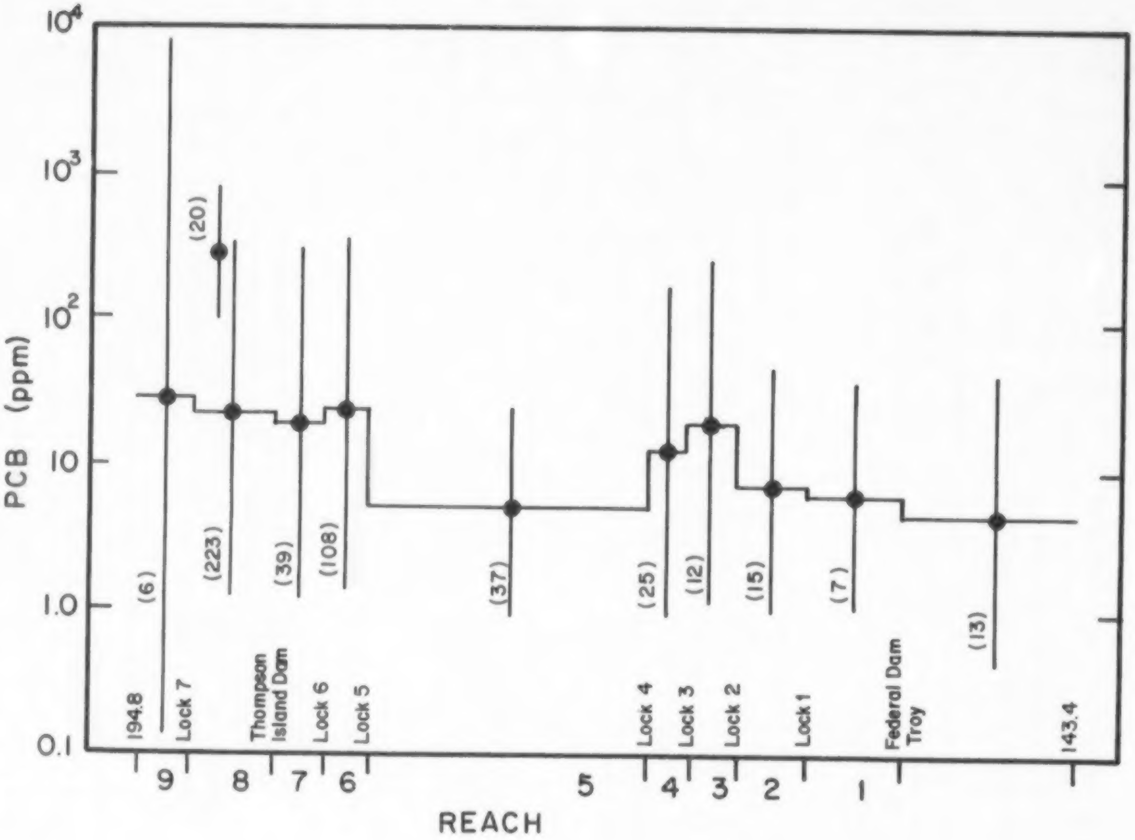

Figure 4. Average PCB concentration in sediments of Hudson River, log mean and 95% confidence interval (number of samples)

DEPTH IN CORE (INCHES)	REACHES 8,9			REACHES 6,7			REACHES 1-5		
	FREQUENCY OF PCB>50 ppm	MEAN PCB (ppm)	N	FREQUENCY OF PCB >50 ppm	MEAN PCB (ppm)	N	FREQUENCY OF PCB > 50 ppm	MEAN PCB (ppm)	N
3	.39	93	87	.40	102	47	.08	22	48
6	.58	112	57	.57	153	23	.20	47	25
12	.36	120	103	.29	130	51	.29	49	62
18	.33	133	63	.18	48	22	.05	14	22
24	.25	75	27	0	6.3	11	0	5	6
36	.22	69	9	0	3.0	1	0	3	1
	0	4.7	7						

Figure 5. PCB penetration in cores by reach from areas where PCB > 5 ppm.

1.5 to 2.5 cm of sediment deposition per year on the average, and this placed the peak PCB deposition period in the 1960s. Cs 137 followed a log normal distribution in the sediments, so log Cs 137 and log PCB were related for all 12 winter cores with a resulting correlation coefficient of 0.82. Certain heavy metals concentrations and percent silt were also positively correlated with PCB (9).

As a result of the data analysis, the Upper Hudson Riverbed sediments were divided into two areas, the so called "hot spots" (PCB concentration > 50 ppm) and "cold areas" (PCB concentration < 50 ppm). The total mass of PCBs in each of these general areas was calculated and is presented in Table 5.

TABLE 5. ESTIMATED MASS OF PCB IN SEDIMENTS OF UPPER HUDSON RIVER

	PCB mass (thousands of lbs)			
Reach	Total	Hot Spot	Cold Area	Scourable*
9	3	**	3	2
8	118	98	20	47
7	16	6	10	4
6	49	41	8	9
5	45	11	34	9
4	20	12	8	7
3	18	2	16	11
2	14	0	14	3
1	13	0	13	6
All	296	170	126	98

Sources: Tofflemire and Quinn (10).

* Estimated from relative scour velocities determined from sediment transport model.

** The average concentration for this reach is 20 ppm, but sampling is inadequate to define hot spots and cold areas. It should probably be considered a cold area.

MANAGEMENT ALTERNATIVES

During the initial stages of formulating the plan of study for the PCB Hudson River Project, a number of potential methods for managing the PCB contaminated sediments were reviewed. Some of these methods consisted of: degradation by ultra-violet ozonation, biodegration, chemical treatment, adsorption, chemical fixation, covering PCB contaminated sediments, dredging, bioharvesting, etc. Most of these alternatives could not be applied to the Hudson River problem because of the high cost and/or impracticability. Others were judged not to be sufficiently technically developed for possible use in the Hudson River Project.

The development could require a 5 to 10 year period before a full evaluation could be made. In the meantime, contaminated sediments would continue to move into the estuarine portion of the river and possible major flood conditions could further disperse the known areas of high contamination. Because of the urgency in the nature of the project, the major management alternative that was studied in detail was dredging.

Four alternatives which appear feasible are summarized in Table 6. The following is a brief description of the four alternatives and the effectiveness of each in removing PCBs from the river.

No Management Action (Maintenance dredging)

If no action takes place in the river, New York State Department of Transportation, empowered by the State's constitution, has the task of maintaining the State canal system. Based on past requirements to maintain a navigable, 12 foot deep channel in the Upper Hudson River, it has been estimated that DOT will be required to remove over 900,000 yd^3 of sediments over the next ten years.

This maintenance program will in fact remove approximately 50,000 pounds of PCBs from the river. This estimate was made on the assumption that the removal sediments would contain about 35 ppm of PCB and that a 90 percent removal efficiency could be achieved under a carefully controlled dredging program.

Remnant Deposits

The remnant deposits are those formed by the sediments that settled behind the former Fort Edward Dam. These deposits can erode easily, especially during high river flows. They cover five distinct areas in a 2 mile stretch of river. Three of the areas contain a low PCB concentration of 1-20 ppm, and no structural action was planned. The remaining two areas, although high in PCBs (25-200 ppm), were stabilized and could be removed later if necessary. An additional area which contained an average PCB contamination of 1000 ppm, was removed and placed in a secured, encapsulated facility. The excavation of two highly contaminated areas would eliminate about 140,000 lbs of PCBs.

Hot Spot Dredging

The greatest amount of PCBs can be removed economically by dredging the hot spot areas. The main reason for considering the hot spot dredging was the fact that the unit costs ($/pound PCB) of removal was the lowest, yet the least environmental disruption would occur. It has been estimated that an efficient, well operated and modernized dredging operation can remove effectively over 94 percent (11) of the PCBs contained in the hot spot areas.

While disturbing only 8 percent of the riverbed area in the Upper Hudson, the hot spot program can remove 160,000 pounds of PCB. Careful evaluation of the various physical conditions in each of the areas will provide the necessary information so that the best equipment for each of the areas can be se-

TABLE 6. SUMMARY DESCRIPTION AND EVALUATION OF POSSIBLE MANAGEMENT STRATEGIES FOR THE HUDSON RIVER

Possible Management Action	Yd^3 Dredged from River (1978-1988)	Lbs. PCBs Removed from River System (1978-1988)	Types of Technology	Status
1. Further study - No management action	916,000	50,800	Modified DOT maintenance dredging program to assure permanent removal of >90% of PCBs dredged from river as part of maintenance program.	Continuous DOT maintenance dredging necessary for Barge Canal operation.
2. Stabilize and/or remove remnant deposits.	1,082,000* (888,000)**	165,000* (97,500)**	Bank stabilization and/or remove remnant deposits and modified DOT maintenance dredging program	Environmental assessment prepared; 2-year program anticipated as outlined above (1978-1979).
3. Remove remnant deposits and remove all river sediments with PCB concentrations greater than 50 ppm (hot spot dredging).	1,660,000	313,000	Removal of remnant deposits and hot spot dredging technology to vary from small specialized hydraulic dredges to large clamshell dredges. Maximum practical PCB containment technology would be used.	Remnant deposit environmental assessment prepared; 2-year program anticipated; hot spot environmental assessment under preparation; 3-year program anticipated (1980-1982).
4. Removal of all river deposits greater than 1 ppm (>5 ppm in remnant deposit areas).	15,000,000	410,000	Massive clamshell dredging effort. Maximum practical PCB containment technology would be used.	Environmental assessment not as yet authorized; 10-year program anticipated (1980-1990)

Source: calculated from Malcolm Pirnie (7, 14) and Tofflemire and Quinn (10).
* Assumes area 3, 3A and 5 are removed from the basin.
** Assumes area 3A and 5 are removed and area 3 is stabilized.

lected. Much more sampling and related geological and engineering work must be done in order to fully delineate the hot spot areas, which could be fully dredged within a 2 to 3 year period.

Complete Dredging

The most expensive and extensive dredging program could be undertaken if funds were available for dredging the entire shore-to-shore 40 mile section of the Upper Hudson River. The dredging yardage involved in this program would be an order of magnitude greater than the hot spot dredging, but it would allow well over 94 percent of the total PCBs to be removed from the riverbed between Fort Edward and Troy, New York.

This removal project, because of its extensive scope of work, would require approximately 10 years to complete.

DISPOSAL

A number of methods for final disposal were evaluated. Among these were incineration, biodegradation and total encapsulation.

Incineration

The technical, economic and environmental feasibility of destroying PCBs in river sediments was investigated by GE (12). As long as temperatures exceeded 1800°F (982°C) and 0.5 seconds contact time, most PCBs were destroyed.

The incineration process is expensive, however, it may become attractive if a co-disposal or tri-disposal system could be designed. That is, the incineration system could provide ultimate disposal for both the solid waste and wastewater sludge stream as well as toxic waste, such as PCBs in river sediments. Disposal schemes, such as the one being developed by Wright-Malta Corporation (13), that is, a gasifier-gas turbine system, could generate electric power to off-set the yearly operation and maintenance costs.

Biodegradation

Biodegradation is a possibility but sufficient information does not now exist to design such a system. Environmentally and economically, it is a very attractive and desirable option that has not yet been technically developed.

Total Encapsulation

This process involves the total containment and isolation of contaminated material within a land burial facility. The Federal and State governments have issued very extensive regulations regarding the design and operation of toxic chemical landfills, especially with regards to PCBs. These criteria relate to site selection, lining and leachate prevention, and operation and maintenance.

Detailed information (7) was developed for each of the criteria. Over 100,000 acres of land, along a 2 mile corridor on both sides of the river between Fort Edward and Troy, were evaluated (14). Forty potential sites were selected during the preliminary period and now six remaining candidate sites are being fully evaluated before a decision is made. Detailed field studies are now in progress in order to select the best possible site and one that lends to the dredging program.

Potential for environmental losses will be minimized by placing an impermeable barrier beneath the contaminated sediments and a similar impermeable cap above it to prevent water from infiltrating into the dredged material.

Long term operation and maintenance provisions must be made in order to guarantee the impermeability of the cap, the collection and disposal of leachate, if any, and a complete environmental monitoring program.

New York State as well as the rest of the country has little experience in many of these areas, especially in the design, construction, operation and maintenance of an encapsulated toxic substance facility. Pilot projects and other dredging operations presently on-going in New York State will facilitate the final design of PCB dredging and disposal program.

ENVIRONMENTAL IMPACTS

The public health impact of the PCBs in the sediments of the Hudson River is a primary reason for pursuing the removal project. Secondly, the fishery resource for most of the river is unavailable to New York residents and visitors.

Whether one is concerned with public health or ecological benefits, the remedial project for the Hudson River will have a definite impact on the PCB contamination of fish. Any action that can economically remove and isolate a toxic material from the environment should be given high priority.

A summary of both short and long terms effects of the alternatives that were considered are presented in Tables 7 and 8.

TABLE 7. SUMMARY OF SHORT TERM ENVIRONMENTAL IMPACT OF VARIOUS MANAGEMENT ALTERNATIVES

	Aesthetic	Biota	Water Supply	Land
1. Further study, No management action	Minimal disturbance in sections of river dredged as part of DOT maintenance dredging program.	Localized elimination of benthic fauna in maintenance dredging areas.	Minimal increase in PCBs in vicinity of Waterford water supply intake when areas adjacent to water intake are dredged.	Land areas involved are small and land based operation will be local and of minimal environmental significance.
2. Stabilize/remove remnant deposits	Extremely localized.	Temporary increase of PCB levels in macroinvertebrates immediately downstream of dredging.	No short term impact other than as stated above for maintenance dredging.	2 - 8 ha of land may be completely disrupted.
3. Remove remnant deposits plus hot spot dredging	Local water quality standard for turbidity will be violated during dredging operation (2-3 yrs).	Temporary (1-2 years) loss of benthic fauna in dredged hot spot areas. Short term rise in PCB concentration in biota immediately downstream from dredging area. Temporary increase of PCB levels in macroinvertebrates immediately downstream of dredging. Loss of wetlands.	Since there are no hot spots in vicinity of Waterford intake, no water supply impacts are expected other than the maintenance dredging impact stated above.	30 - 70 ha of land will be completely disrupted.
4. Removal of all river deposits with PCBs greater than 1 ppm (5 ppm in remnant deposits)	Potentially extensive. Continued local disturbance in the areas under active dredging will cause aesthetic problems for 8-10 yrs.	Temporary elimination of benthic fauna in dredged areas for period of up to 3 yrs after dredging. Temporary increase of PCB levels in macroinvertebrates immediately downstream of dredging.	Temporary increase in PCB concentrations in vicinity of Waterford water supply intake when that segment of the river is dredged.	Up to 400 ha of land will be disrupted.

TABLE 8. SUMMARY OF LONG TERM ENVIRONMENTAL IMPACT OF VARIOUS MANAGEMENT ALTERNATIVES

	PCB Containment	Biota (11)	Water Supply	Land	Air
1. Further study - No Management action	Approximately 23,000 kg of the PCBs will be recovered from the riverbed by maintenance dredging, leaving 175,000 kg (or 88% of the inplace upper river PCBs) to disperse and probably recycle through the environment.	PCB levels in fish and other biota in the upper and lower river will not change in the foreseeable future (upper river >10, lower river >20); lower river concentrations may rise as highly contaminated PCB sediments move from the upper river into the estuary.	Water column PCB concentrations will continue to cause public health concerns for people using the river as a source of drinking water. Any plans to use the Hudson for for drinking water in the future will have to accomodate this contamination.	No long term land impact is foreseen.	Continued probable volatilization from the river and dredge spoil from maintenance dredging.
2. Stabilize/remove remnant deposits	As much as 75,000 kg of of PCBs will be permanently removed from the riverbed leaving 122,000 kg (or 63% of the upper river inplace PCBs) to disperse and continually recycle through the environment. Stabilization of the remnant deposits will substantially eliminate a known active source of PCBs to the upper river system; will reduce size and PCB complexities of of DOT maintenance dredging program.	Will reduce by an unknown amount the time required for PCB levels in the upper river biota to fall. Minimal effects on the lower river biota are anticipated.	May reduce water column PCB concentrations in in vicinity of Waterford water intake by reducing an active source to the upper river. Will have only a minimal effect on any lower river water supplies, and planned and planned flood skimming for NYC water supply in the foreseeable future.	2 - 8 ha of land will be required for spoil disposal. Permanent restrictions on use of the land will be required. Restrictions may be mitigated by using land as a nature preserve or ecological study area. (Permanent restrictions could be changed by technological advancement in PCB bacterial, biodegradation research).	Reduction of volatilization from river probably slight.
3. Remove remnant deposits and remove all river sediments with PCB concentrations >50 ppm. (hot spot dredging).	Approximately 142,000 kg of PCB will be permanently removed from the riverbed leaving 56,000 kg (or 28% of the upper river inplace PCBs) to disperse and continually recycle through the environment. This option will eliminate all high concentration sediments (>50 ppm) from the river system.	Can reduce PCB biota concentrations in upper river by 50% to fish flesh levels of 20-40 ppm. In the lower river, fish burdens may drop by about 20% to about 8-12 ppm.	Should substantially reduce water column PCB concentrations in vicinity of Waterford water supply intake. Will have a marginal effect on lower river water supplies, existing and planned in the foreseeable future.	30 - 70 ha of land will be required for contaminated spoils disposal. Permanent restrictions on use of land will be required. Restrictions may be mitigated by using land as a nature preserve or ecological study area. (Permanent restrictions could be	Should substantially reduce PCB volatilization from the upper river.

TABLE 8. (continued)

	It will substantially reduce the size of PCB complexity of DOT maintenance dredging operations. The sediments to be recovered are also generally contaminated with organic pollutants and other toxic materials. This project would remove these materials from the river environment.			changed by technological advancement in PCB bacterial, biodegradation research).	
4. Remove all upper river deposits >1 ppm (>5 ppm in remnant deposit areas).	About 186,000 kg (~ 94%) of the PCBs in the upper river will be permanently removed from the riverbed and the environment. Substantial reductions in the size and PCB complexities of future DOT maintenance dredging programs will occur. Most of the sediments to be removed are also contaminated with other organic pollutants and other toxic materials. This project will remove these materials from the river environment.	Can reduce PCB biota concentrations in the upper river to fish flesh levels of 10-20 ppm. By significantly reducing PCB input into the lower river, it may reduce the body burden of larger fish by about 30-50% to levels close to 5 ppm.	Will substantially eliminate the problem of PCBs in the village of Waterford water supply intake. Will also lower level of other toxic contaminants in the Waterford water supply. Can measurably reduce PCB water column problems in vicinity of estuary water supply intakes by stopping active sources of PCBs to the lower river. May reduce PCB problems associated with flood skimming project.	About 400 ha of of land will be required for contaminated spoil disposal. Permanent restrictions on use of land will be required. Restrictions may be mitigated by using land as a nature preserve or ecological study area. (Permanent restrictions could be changed by technological advancement in PCB bacterial, biodegradation research).	Will eliminate the river as a source of PCBs to atmosphere. Odors associated with extensive sludge beds should also be essentially eliminated.

FUNDING

A number of funding methods have been considered in order to complete the Hudson River remedial program. These included:

1. State/local funding.

2. Funding as a Corps of Engineers (COE) public works project.

3. Funding by special Congressional legislation.

4. Funding under the Water Pollution Control Act. (Clean Waters Act).

New York State is not able to finance the cost of this remedial program. The Federal government together with New York State should bear the burden of financing the large remedial program, since the entire nation has benefited by the capacitor manufacturing process at GE's Fort Edward and Hudson Falls plants.

Funding both as a Corps of Engineers public works project or through special legislation would require a long period of time and unacceptable delays. Contamination of the lower Hudson continues at a rate of over 5,000 lbs per year and the cost of the project also continues to rise.

New York State has seized the opportunity to fund the remedial project under the Federal Water Pollution Control Act. Two portions of the Act that appeared relevant were a research and development grant under Title I and a treatment works construction grant under Title II.

Two grants requests were submitted to EPA under Title I, Section 115 and under Title II, Section 201. Innovative and alternative wastewater treatment techniques are specifically encouraged by Section 201(g)(5) of the Water Pollution Control Act for the purpose of confining disposal of pollutants, so that pollutants will not migrate and cause water or other environmental pollution.

To this day, New York State has not received eligibility ruling for either of these grant applications. EPA in Washington is presently reviewing the project under Section 201 and insufficient funds exist in Section 115.

Two additional areas of funding are being pursued: a) a special Congressional action on Clean Waters Act amendment to authorize the PCB project (proposed New Section 116), and b) the Superfund Bill that is now in Congress. Both of these areas could produce funds by spring of 1980. New York State is fully committed to this project and is continuing to obtain additional scientific and engineering data so that it can be in a position to start the PCB remedial project in 1980.

STATUS

The New York State Department of Environmental Conservation through a small but highly technical staff is continuing to finance the on-going scientific research and has begun to formulate, in detail, the management schedule for the implementation of the PCB Reclamation Project. It is a difficult task and there are many steps that could delay the project (Figure 6). However, the commitment to eliminate the spreading contamination of the Hudson River by PCB has long been made by the State of New York. Some initial phases of this project were already taken in the summer of 1978 (15). New York state Department of Transportation, as part of the maintenance dredging, has removed an additional 7,000 lbs of PCBs in the summer and fall of 1979. And, if the funding becomes a reality by spring of 1980, it is possible that the PCB Hot Spot Dredging Project for the Hudson River could be completed by 1982.

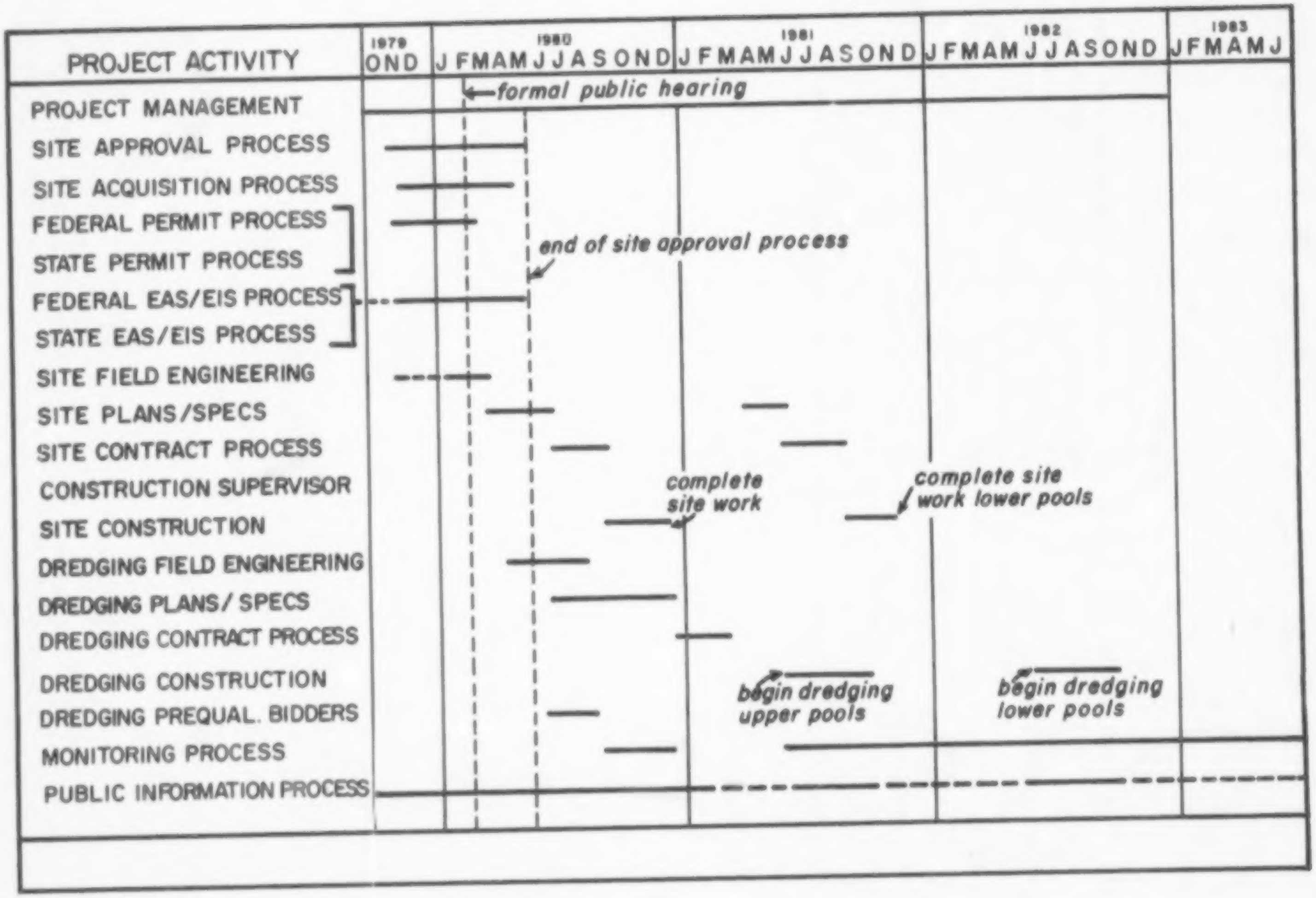

Figure 6. Hudson River PCB reclamation project - accelerated schedule.

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THE SECTION 404 DREDGE AND FILL PROGRAM

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ABSTRACT

Dredging and filling in the navigable-in-fact waters of the United States have been regulated for 80 years under provisions of section 10 of the River and Harbor Act of 1899. Since 1968, major federal environmental legislation has expanded the criteria for decision on section 10 permit applications to include numerous public interest factors in addition to navigation. With enactment of the Federal Water Pollution Control Amendments of 1972, section 13 of the 1899 Act was amended and expanded to create a water pollution control program regulating discharges of all classes of pollutants into the Nation's waters. The new statute established the section 404 program to regulate discharges of dredged or fill material, with responsibilities for administration divided between the department of the Army and the Administrator of the Environmental Protection Agency. The geographic scope of the section 404 program is substantially broader than that available in the 1899 Act, but the classes of activity regulated are to some degree similar.

Environmental criteria for the Section 404 program are based on objectives that encourage minimization of impact principally through careful evaluation of material proposed for discharge and through rigorous evaluation of alternative disposal sites and methodologies.

INTRODUCTION

The section 404 permit program for the regulation of discharges of dredged and fill materials in waters of the United States under an authority directed specifically and exclusively to that purpose is of relatively recent origin. A closely-related regulatory program has been carried out for 80 years under section 10 of the River and Harbor Act of 1899. The interrelationships between these programs are so strong that a discussion of the one is incomplete without due reference to the other. The purpose of this paper is to describe briefly the legislative origins of the section 404 program; the geographic scope and range of activities regulated by the program; and the environmental review criteria used to review applications for section 404 permits.

HISTORY OF FEDERAL LEGAL AUTHORITIES AFFECTING DREDGING AND DREDGED MATERIAL DISPOSAL

River and Harbor Act of 1899

The principal authorities for the regulation of dredging and the discharge of dredged material in waters of the United States are section 10 of the River and Harbor Act of 1899 (33 U.S.C. 401 et seq.) and section 404 of the Clean Water Act (13 U.S.C. 1344). The first of these authorities essentially constitutes an assertion of police power to protect from damage or obstruction those waters which are in use for the transportation of interstate or foreign commerce, which have been used for such purposes in the past, or which, with reasonable improvement, could be made usable for that purpose.

Section 10 prohibits the construction or placement of any structure in navigable waters of the United States unless authorized by a Department of the Army permit. This includes docks, piers, wharves, bulkheads, weirs, booms, breakwaters, jetties, pilings, power transmission lines, cables, aids to navigation, and similar structures. Section 10 also requires a permit for such work in navigable waters as dredging, disposal of dredged material, filling, and the connection of artificial canals and ditches to existing navigable waters in any manner which might affect their navigable capacity.

For about the first 69 years of its implementation, the 1899 Act was interpreted and implemented exclusively in terms of navigational concerns, that is, the sole criterion for issuance or denial of a permit was the extent to which proposed structures or work would affect navigation. In 1968, the Department of the Army revised the regulations implementing section 10 (and other authorities under the 1899 Act) to expand the scope of permit review to include the effect of proposed work or structures "... on navigation, fish and wildlife, conservation, pollution, aesthetics, ecology, and the general public interest ..." (33 CFR 209.120(d), 1968). In the following decade, a number of new environmental laws were enacted, the effect of which was to increasingly expand the environmental factors taken into account in the review of section 10 permit applications. Among the major laws enacted in this period were:

> The National Environmental Policy Act of 1969 (PL 91-90, 42 U.S.C. 4331 et seq.), which requires Federal agencies to evaluate the environmental consequences of their actions. For major actions that significantly affect the environment, agencies are required to develop detailed "environmental impact statements" and to submit these to interagency and public review before making a final decision of whether to carry out a proposed agency action.

The National Historic Preservation Act of 1966 (80 Stat. 915, 16 U.S.C. 470), which established a National Advisory Council on Historic Preservation to advise the President and the Congress on matters of historic preservation and to review and comment on activities licensed or permitted by the Federal government which will have an affect on major historical properties.

The Coastal Zone Management Act of 1972 (PL 92-583, 86 stat. 1280), which establishes procedures for States to follow in development and operation of Federally-approved coastal zone management programs. These programs involve the development of comprehensive plans for the long-term protection and development of coastal land and water resources. The Act requires any applicant for a Federal permit or license to conduct any activity affecting coastal land or water uses to furnish the permitting or licensing agency with a certification that the proposed activity complies with the State's coastal zone management program. Generally, no Federal permit or license will be issued for such an activity until the State concurs with the applicant's certification.

I have cited these particular authorities because they are major environmental laws which, in combination with the River and Harbor Act of 1899, have substantially improved the degree to which environmental values are protected through the Department of the Army's section 10 permit program. These statutes, together with a number of other environmental and other laws, have led the Department of the Army, acting through the U.S. Army Corps of Engineers, to expand the public interest review of permit applications to embrace, currently, a total of 17 specifically named factors. These are, "conservation, economics, aesthetics, general environmental concerns, historic values, fish and wildlife values, flood damage prevention, land use, navigation, recreation, water supply, water quality, energy needs, safety, food production and in general the needs and welfare of the people" (33 C.F.R. 325.3(b)(1)).

Federal Water Pollution Control Act Amendments of 1972

By the year 1972, the scope of the public interest review of permit applications under the 1899 River and Harbor Act was commendably broad. Although that Act itself was not designed or implemented primarily for environmental protection, it was evolving into an effective tool for that purpose when applied in combination with other laws. In 1972, widespread public concern about the degraded condition of the Nation's waters prompted the Congress to consolidate, revise, and expand upon the existing body of Federal law controlling water pollution. The product of these Congressional labors was a comprehensive water pollution control act, entitled the Federal Water Pollution Control Act of 1972 (FWPCA) (33 U.S.C. 1344), which embodies a national goal "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters".

The 1972 Act did not change section 10 of the River and Harbor Act of 1899. It did, however, amend another Section of that Act, namely section 13.

Section 13, referred to as the "Refuse Act," prohibited the discharge of refuse mater (except liquid sewage) into any navigable water of the United States or any tributary thereof unless authorized by a permit from the Secretary of the Army and the Chief of Engineers. For many years, Section 13 lay in almost total disuse until, in the mid-1960's, it reemerged as the primary Federal enforcement tool for control of pollution discharges into the Nation's navigable waters. Section 402 of the FWPCA essentially superseded section 13 by transferring to the Administrator of the Environmental Protection Agency the permitting authority for discharge of most classes of point source pollutants.

The early drafts of the FWPCA envisioned a single permit program for the regulation of point source discharges of all pollutants into the navigable waters, with authority to administer this program to be vested with the Administration of the Environmental Protection Agency. As the 1972 legislation evolved within the Congress, several considerations emerged which impelled the Congress to the eventual establishment of a separate permit program to control the discharge of dredged or fill material. First, the Congress recognized that management of dredged material was a responsibility that had long been vested in the Secreatry of the Army through section 10 of the River and Harbor Act of 1899 and that for this reason an established administrative system was already in place, within the U.S. Army Corps of Engineers, for the processing of dredge and fill permit applications. For this reason, the Congress "did not wish to create a burdensome bureaucracy in light of the fact that a system to issue permits already existed."(1) It was also argued, by certain elements of the Senate, that a regulatory program so vitally tied to the economics of navigation should not be entrusted to a newly-established regulatory agency such as EPA, which, at the time, had been in existence for only about two years. The outcome of the Congressional deliberations on this issue was to establish separate permit programs for the regulation of point-source pollutants. Section 402 of the FWPCA provides a permit program, characterized as the National Pollutant Discharge Elimination System, to regulate point souce discharges of municipal and industrial point source discharges of water pollutants. The Administrator of the Environmental Protection Agency is responsible for the administration of this program. Section 404 of the 1972 Act, and as subsequently amended, provides a permit program exclusively for regulation of discharges of dredged and fill material. The Secretary of the Army, acting through the Chief of Engineers, carries out the permit program under this section. (A 1977 Amendment provides that individual States may operate their own section 404 programs under standards prescribed by law, after approval by the Administrator. State programs, however, cannot regulate discharges into waters actually used for transportation of waterborne commerce or capable of being put to such use.) The Administrator of the Environmental Protection Agency, however, is provided a very substantial role in the section 404 program in that he is required to develop guidelines to be used, by the Corps and by States, in the evaluation of all proposed discharges of dredged or fill material. The Administrator is also required to monitor the operation of approved State programs and can revoke State programs where he finds that they are not being carried out as required by law. Moreover, the Administrator is provided (at section 404(c)) with the authority (frequently alluded to as a "veto power") to restrict or deny the use of any site as a site for the discharge of dredged or fill material. This means that even in cases where the Chief of Engineers or a State official has issued or would issue a permit, the Administrator can, for sufficient cause, use his authority to prevent the discharge from being made. In explaining the basis for this unusual balance of powers in Section 404, the Conference Committee stated that, it:

> "... did not believe that there could be any justification for permitting the Secretary of the Army to make determinations as to the environmental implications of either the site to be selected or the specific spoil to be disposed of in a site. Thus, the conferees agreed that the Administrator of EPA should have the veto over the selection of any site for dredged spoil disposal and over any specific spoil to be disposed of in any selected site" (1).

In summary, then, in the Federal Water Pollution Control Act Amendments of 1972, the Congress formally recognized dredged material (as well as fill material) as a pollutant; acknowledged the substantial experience of the Corps of Engineers in managing these materials via the section 10 permit program and assigned administration of the section 404 permit program to the Corps; and provided the Administrator of EPA with substantial authorities designed to assure that this new program would focus effectively upon its primary purpose of controlling water pollution.

SCOPE OF THE CURRENT SECTION 404 PROGRAM

Geographic Scope

As explained above, the 1899 River and Harbor Act is applicable only to those waters that are, have been, or could be used for the actual transportation of interstate commerce. For the first several years of its administration, the Corps of Engineers administered the section 404 program only within these same waters. A number of private environmental organizations opposed the Corps' interpretation of this point. They agreed that the Congress intended, in section 404 and in the rest of the 1972 Act, to control pollution in all of the waters of the United States where discharges of pollutants could affect the quality and value of those waters for any purpose of interstate commerce, not merely for navigation. In 1974, the Natural Resources Defense Counsel and the National Wildlife Federation filed suit against the Secretary of the Army, the Corps of Engineers, and the Environmental Protection Agency* (NRDC v. Callaway, 392 F. Supp. 685 D.D.C. 1975). These plaintiffs asked the court to compel the Department of the Army to extend its section 404 jurisdiction to all waters, the pollution of which could affect interstate commerce. This litigation was successful, and on March 27, 1975, the District Court for the District of Columbia ordered the Department of the Army to revoke and rescind that part of its regulation which limits jurisdiction of the Corps by definition or otherwise to other than the waters of the United States." On March 27, 1975, the court directed that the Department revise its regulation to extend the geographical jurisdiction of the section 404 program to the maximum extend permissible consistent with the commerce clause of the United State Constitution. The current Corps of Engineers permit regulation responds to this mandate. Accordingly, the jurisdiction of section 404 extends into all waters subject to use for navigation; recreational use; production of fish and shellfish; production of agricultural products; industrial water supply; or for any other purpose which involves the use of the waters for interstate or foreign commerce purposes.

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* In addition to seeking an expansion of geographical jurisdiction for the section 404 program, the plaintiffs sought to compel EPA and the Corps to publish the section 404(b)(1) guidelines which, although required by statute, had not been published at the time the suit was filed.

This broad jurisdiction means that section 404 regulates activities in many waters of the United States that never were subject to regulation under the section 10 permit program. Section 10, for the most part, extended only to coastal waters and the inland waters that did or could actually support navigation. There are many thousands of miles of streams within the United States and millions of acres of standing water bodies that are incapable of supporting navigation, but that are of extremely high importance for fish and shellfish production; wildlife habitat; municipal and industrial water supply; timber production; recreation; and other purposes having connection with interstate commerce. Most notably among these are wetlands, which are typically flooded for only a portion of the year, but which perform a number of ecologically and socially valuable functions, including fish and wildlife production; erosion control; filtration and purification of runoff waters; temperature stabilization; and flood storage. It is estimated that there may be as many as 37 million hectares of wetlands within the United States, excluding Alaska and Hawaii (2). Prior to the decision in NRDC v. Callaway, the only wetlands regulated under section 404 were coastal wetlands subject to the regular ebb and flow of the tide and freshwater wetlands lying below the ordinary high water mark of navigable-in-fact freshwater bodies. The extension of section 404 jurisdiction to most of the wetlands was widely applauded by environmental interests who had long been concerned about the widespread and indiscriminate filling of wetlands and the very substantial public values lost through such actions.

Also of great significance is the extension of section 404 jurisdiction to small streams, including those streams that support runs of anadromous fishes such as salmon, steelhead, striped bass and other fish species of major importance to domestic and international sport and commercial fisheries.

Workload Implications

This manyfold expansion of jurisdiction could easily have created burdensome administrative problems in the processing of the numerous additional permit applications that resulted, and in the task of enforcement against unpermitted discharges of dredged or fill material into the newly-regulated waters. These problems were largely avoided by several means, including: (1) a nationwide publicity campaign, including public hearings and public meetings, to inform the public of the extended jurisdiction; (2) the development of a "general permit" program. General permits authorize minor discharges of dredged or fill materials, to be undertaken under the terms of an already-issued permit. A general permit specifies the conditions that must be met by a discharger in order to be in compliance with the permit. Once a permit has been issued, anyone desiring to perform work authorized by the general permit may proceed to do so, without being required to make an individual application and without having to wait for a permit application to be processed. This is a major savings of time, since the minimum period for obtaining individual section 404 permit application is about 45 days from the date an application is submitted.

To date, over 200 general permits have been issued by District Engineers of the U.S. Army Corps of Engineers. In addition, two nationwide general permits were issued in the Corps of Engineers' most recent permit regulation (33 C.R.F. 323.4). The general permit program has probably authorized tens of thousands of individual minor discharges of dredged or fill material into the waters of the United States. (3) increased Department of the Army staff to process permit applications. The Congress, recognizing the need for additional personnel to operate the expanded section 404 program, increased the staff in the Corps of Engineers regulatory program to over 1,000 persons.

The Environmental Protection Agency has also increased the number of personnel assigned to carry out its responsibilities in the section 404 program. EPA had fewer than 30 positions assigned to the program in 1975. Total EPA personnel now numbers 62.

Scope of Activities Regulated

Dredging not regulated

The section 404 program is frequently characterized, both coloquially and formally, as a "dredge-and-fill" program. The term was widely used to describe activities regulated under the section 10 permit program. That program did, in fact, regulate the act of dredging, or excavation, of material from ports, harbors, channels, and other navigable-in-fact waters. By contrast, section 404 is intended only to regulate actual discharges of dredged or fill materials and therefore does not regulate the act of dredging itself.

The distinction is not unimportant, for it is entirely possible for a dredging operation to be carried out in a manner which does not involve the discharge of dredged material into the waters of the United States. No section 404 permit is required to authorize the discharge of dredged material that is conveyed to, and deposited upon a dry-land disposal site, but the environmental standards by which section 10 applications are evaluated are not as strict as those under section 404. If the dredging is carried out in waters not subject to section 10, and the discharge is confined entirely to dry land, the dredger avoids regulation under either authority. For these reasons, neither section 10, nor section 404, nor the two laws working in concert can be considered to be comprehensive in their regulation of dredging and filling in all waters of the United States. These relationships are shown graphically in Table 1.

TABLE 1. COMPARISON OF THE GEOGRAPHICAL REACH OF TWO FEDERAL REGULATORY PROGRAMS AND THEIR EXTENT OF REGULATION OF THE ACTS OF DREDGING AND FILLING

	Type and Location of Activity			
Statutory Authority	Dredging in navigable-in-fact waters*	Dredging in other than navigable-in-fact waters†	Discharge of dredged/fill material in navigable-in-fact waters	Discharge of dredged/fill material in other than navigable-in-fact waters
Section 404 of Clean Water Act	not regulated	not regulated	regulated	regulated
Section 10 of River and Harbor Act of 1899	regulated	not regulated	regulated	not regulated

* "navigable-in-fact waters" means coastal waters that are subject to ebb and flow of the tide shoreward to the mean high water mark (mean higher high water mark on the Pacific coast), and/or are presently used or have been used in the past, or may be susceptible to use to transport interstate or foreign commerce.

† "other than navigable-in-fact" includes waters other than "navigable-in-fact waters" the degradation or destruction of which could affect interstate commerce.

Refuse Discharges Not Regulated

Except for dredged material, section 404, as currently implemented does not regulate the discharge of any form of waste or refuse. Fill material, the second category of pollutant regulated under section 404, is defined, for regulatory purposes, as "any material used for the primary purpose of replacing any water of the United States with dry land or of changing bottom elevation of a waterbody". "Dredged material" is defined as "material that is excavated or dredged from waters of the United States". Considering these two regulatory definitions together, it is interesting to contemplate a situation in which soil, clay, sand, rock, or some other earthern substance is excavated from an area outside the waters of the United States, but is then discharged into waters of the United States for the purpose of getting rid of it. The material handled can not be defined as "dredged material", since its origin is outside the waters. Nor would it meet the definition of "fill material", since it is not discharged for either of the two purposes which that definition requires. It may be possible to close this unfortunate loophole in the coverage of section 404 by revising the definition of fill material to expand the purposes associated with that term. Unless and until that is done, such discharges can be regulated under section 402 of the Act. The section 402 program, however, has not been administratively designed to handle these kinds of discharges, and thus is ill-equipped to do so.

Range of Activities Regulated

Two district classes of activity are regulated by the section 404 program. Discharges of dredged material, for the most part, are undertaken in conjunction with navigation. In the United States, annual quantities of materials dredged by or in behalf of the Corps of Engineers average about 290,000,000 cubic yards for maintenance dredging and 78,000,000 cubic yards for new work dredging (3). Additional navigation dredging is performed by other interests, both public and private, but the vast majority of all dredging for navigation in United States waters is carried out by the Corps. Typical activities of the Corps are construction and maintenance dredging of channels, canals, turning basins, small boat harbors, and ocean inlets.

Private industry dredging, and associated disposal, typically includes the dredging of ship berths; marinas; residential canals; canals and slips for movement of equipment used in exploration and extraction of oil, gas, sulfur and other minerals; and dredging of shell, gravel and sand for manufacture of construction materials or roadway construction.

As discussed above, it is possible to discharge dredged material on dry land sites and to thereby avoid regulation under section 404. In actual practice, however, a large proportion of the material dredged from the waters of the United States is also discharged into those waters, at other localities. Therefore, in most instances involving major dredging for navigation, a section 404 permit is required for the discharge, as well as a permit under section 10 of the 1899 River and Harbor Act.

Unlike the discharge of dredged material, the discharge of fill material is not dominantly associated with a single agency or activity. Common uses of fill material are the restoration of eroded sand beaches; stabilization of shorelines with riprap; construction of sites for commercial industrial facilities, including power generation plants; construction of wharves, jetties, groins, breakwaters, bulkheads, causeways, roads, bridge abutments, dikes, dams and levees, and burial of utility lines. Much less commonly, discharges of fill material are made for the purpose of constructing residential sites, solid waste containment facilities, raw materials storage sites, and for other purposes that may not intrinsically require proximity to water, but which can not, in particular situations practicably be sited elsewhere.

Volume of Permits Processed

In 1978, the most recent year for which complete records are available, a total of 10,150 permit applications were reviewed under the section 404 program. Of these, some 6,972 involved activities in the navigable-in-fact waters that were also subject to section 10 of the River and Harbor Act of 1899. In such cases, the permit processing procedures currently in use provide for the simultaneous public notice and review of both permit applications. About 9,000 additional permit applications were reviewed in 1978 for activities that were regulated only under authority of section 10 of the River and Harbor Act of 1899.

ENVIRONMENTAL CRITERIA FOR REVIEW OF SECTION 404 PERMIT APPLICATIONS

Relationship to National Environmental Policy Act

As pointed out above, all Federal actions, regulatory or otherwise, are subject to the requirements of the National Environmental Policy Act of 1969. Neither time nor the purpose of this paper warrant a detailed review of the requirements of that Act, except to take note that those requirements are much broader and more general than are the environmental evaluation criteria mandated in the section 404(b)(1) guidelines (40 CFR 230). The guidelines, as published September 5, 1975, define the procedures to be used in determining the extent to which given discharge of dredged or fill material will adversely impact the quality of the waters of the United States and the living systems and beneficial natural uses which such waters support.

Objectives of Guidelines

The guidelines require that, in the review of all section 404 permit applications, a number of objectives be considered in determining whether to permit a proposed discharge. These objectives are to:

"(1) Avoid discharge activities that significantly disrupt the chemical, physical, and biological integrity of the aquatic ecosystem, of which aquatic biota, the substrate, and the normal fluctuations of the water level are integral components;

(2) Avoid discharge activities that significantly disrupt the food chain including alterations or decrease in diversity of plant and animal species;

(3) Avoid discharge activities that inhibit the movement of fauna, especially their movement into and out of feeding, spawning, breeding and nursery areas;

(4) Avoid discharge activities that will destroy wetland areas having significant functions in maintenance of water quality;

(5) Recognize that discharge activities might destroy or isolate areas that serve the function of retaining natural high waters or flood waters;

(6) Minimize, where practicable, adverse turbidity levels resulting from the discharge;

(7) Minimize discharge activities that will degrade aesthetic, recreational, and economic values;

(8) Avoid degradation of water quality as determined through application of §§230.4, 230.5 (b), (c), and (d)."

The Sections of the guidelines referenced in item (8) above deal with (1) evaluation of physical and chemical-biological interactive effects; (2) the comparison of concentrations of pollutants present in material with the permissible concentrations established in the water quality standards prescribed under section 303 of the Act; (3) consideration to determine the size of a disposal site and the conditions of disposal to minimize the possibility of harmful effects; and (4) the prohibition of discharge of fill material having unacceptable quantities, concentrations, or forms of contaminants deemed "critical" by EPA or the Corps of Engineers, except where adequately confined.

In attempting to achieve the eight objectives listed above, the guidelines require that consideration be given to the need for the proposed discharge and the availability of alternative sites and methods of disposal that are less damaging to the environment. The alteratives analysis lies at the very heart of the section 404 permit evaluation, and it is in this analysis that the section 404 program differs most significantly from the section 402 (NPDES) program. In the latter program, a discharger must meet the requirements of "effluent guidelines", which establish the types and concentrations of defined pollutants that he may lawfully discharge. In the section 404 program, by contrast, the material proposed to be discharged may be completely uncontaminated, but its discharge may nevertheless be prohibited because of the acutely destructive impact of discharging the material upon a biologically productive site, especially when there are other, feasible, alternative sites where the discharge can be made at much less expense to the environment. The section 404(b) guidelines explicitly recognize that, "From a national perspective, the degradation or destruction of aquatic resources by filling operations in wetlands is considered to be the most severe impact covered by these guidelines." (40 CFR 230)

Water Dependency

Proposals for filling wetland areas are submitted to the most rigorous scrutiny of any activity evaluated by the guidelines. In order to obtain a permit for filling of wetlands, the applicant must demonstrate that either (a) the activity is "water-dependent," that is, that it requires direct access or proximity to, or location directly within the water in order to carry out its basic purpose; or (b) other site or construction alternatives are not practicable. He must also show that the proposed fill will not cause a "permanent unacceptable disruption to the beneficial uses of the affected aquatic system." These burdens of proof are sometimes difficult to bear, as indicated in recent years by the denial, or radical reduction in scope, of permit applications involving proposed fill material discharges in wetlands. In one especially noteworthy case, the Corps of Engineers denied two permits to a major real estate developer in Florida, whose proposed projects would have resulted in dredging and filling of about 3,000 acres of coastal wetlands and shallows to create canal and residential homesites (4). The permit was denied even though the applicant had already sold most of the property to hundreds of purchasers of individual homesites and had already obtained all required local and state permits. The principal basis for denial was the finding that the proposed development would "constitute an unacceptable adverse impact upon this aquatic resource" and that destruction of environmentally important wetland areas was "contrary to the public interest" (5). The Corps, in this case, concluded that housing, the basic purpose of the project, was clearly not dependent upon a water location. This landmark case has undoubtedly had a major chilling effect upon the plans of other would-be real-estate developers coveting coastal and freshwater wetland locations as sites for waterfront residential property development.

Proposals for discharge of dredged material as a waste disposal activity (as contrasted with discharge of fill material to create fast land sites) are somewhat less rigorously evaluated in terms of impacts on wetlands. The guidelines recognized that dredging, by definition, is an activity intrinsically arising from use of the waters and that the activity of dredging may entail a need for disposal that cannot be practicably met, in a given case, unless a wetlands site is used. The guidelines require, however, that the applicant demonstrate that the site selected is the least environmentally damaging of the practicable alternatives that are available. Even where this is the case, the permit to discharge may still be denied if the sheer magnitude of the enviornmental impact is unacceptably adverse.

Testing and Evaluation

The guidelines provide general guidance as to the chemical and biological testing and evaluation to be required for dredged or fill materials proposed to be discharged into waters of the United States. Standards are specified, in addition, for the exclusion of material from these testing requirements. No testing is required when:

(1) Dredged or fill material is composed predominantly of sand, gravel, or any other naturally occurring sedimentary material with particle sizes larger than silt, characteristic of and generally found in areas of high current or wave energy such as streams with large bed loads or coastal areas with shifting bars and channels;

(2) Dredged or fill material is for beach nourishment or restoration and is composed predominantly of sand, gravel or shell with particle sizes compatible with material on receiving shores.

(3) When:

  (a) The material proposed for discharge is substantially the same as the substrate at the proposed disposal site; and

  (b) The site from which the material proposed for discharge is to be taken is sufficiently removed from sources of pollution to provide reasonable assurance that such material has not been contaminated by such pollution; and

  (c) Adequate terms and conditions are imposed on the discharge of dredged or fill material to provide reasonable assurance that the material proposed for discharge will not be moved by currents or otherwise in a manner that is damaging to the environment outside the disposal site.

A revision to the guidelines has been formally proposed (6), which substantially expands upon the factors that should be considered in demonstrating reasonable assurance of the lack of toxic contamination in materials proposed to be dredged. Once such assurance is obtained, there is no requirement for chemical and biological testing. The factors to be consulted include:

(1) known potential routes of pollution or polluted sediments to the extraction site;

(2) pertinent results of previous tests on the material at the extraction site;

(3) potential for pesticide contamination from land runoff;

(4) information of record on spills or petroleum products or hazardous substances;

(5) information of record indicative of introduction of pollution from industries along potential routes of contamination to the extraction site; and

(6) possibility of presence of natural mineral deposits (e.g., phosphate) which could be exposed by dredging and released into the aquatic environment.

The rationale for preliminary screening via these factors is to avoid costly, routine, and repetitive testing in those numerous cases where there is no significant reason to believe that dredged or fill materials contain toxic contaminants. Where such a finding cannot be reached, testing is required, with the type and extent of testing for each specific case generally being determined by the judgment of the District Engineer. The Regional Administrator of EPA may require specific testing approaches and procedures on a case-by-case basis by specifying to the District Engineer the type of information needed and stating how the results of the analysis will be of value in evaluating environmental impacts. The types of tests generally employed upon dredged material are:

(1) an elutriate procedure, involving the aqueous extraction of soluble chemical species through vigorous shaking of a water-sediment mixture, followed by settling and filtration of the supernatant. The elutriate (i.e., filtered supernatant) is chemically analyzed for constituents deemed important by the District Engineer, after consultation with the Regional Administrator;

(2) a benthic bioassay of the material proposed for discharge, whereby appropriately sensitive and representative organisms are exposed to this material, and acute toxic effects, if any are measured;

(3) inventory of total concentration of chemical constitutents. This whole sediment analysis may be used to compare sediment at the dredging site with sediment at the disposal site;

(4) analysis of biological community structure. This procedure is essentially an inventory and comparison of biological communities colonizing the substrates at extraction and disposal sites, and may be used in defining the existing degree of environmental stress at both sites.

The testing guidance and the range of tests available in the current guidelines are now widely considered to be in need of improvement. EPA and the Corps are developing a much-expanded testing package that will provide a greater variety of tests; much more structured rationale for the determination of which tests to use in particular situations; and improved guidance in the interpretation of test results. The agencies have also agreed to publish a testing methods manual for use by field personnel and contractor personnel retained by the Corp or by applicants to carry out required tests.

SUMMARY

The section 404 "dredge and fill" program is a relatively new water pollution control program having historic roots in the River and Harbor Act of 1899. The adminstrative structure of the program is unusual in that its substantive responsibilities are divided between two agencies, EPA and the Corps of Engineers. The program is extremely broad in its geographical reach, extending into all waters of the Nation having a nexus with interstate or foreign commerce, but is relatively narrow in terms of the specific activities that it regulates. The program employs stringent evaluation criteria, with particular emphasis upon alternative siting and operational means of reducing or avoiding adverse environmental impacts. In appropriate cases, chemical and/or biological testing of material proposed for discharge is required in order to assess the impacts that would result from release of the material at a proposed disposal site. As the section 404 program matures, its essential technical and regulatory literature are also evolving in the attempt to increase the overall quality and efficiency of program operation.

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SEDIMENT PROBLEMS AND LAKE RESTORATION IN WISCONSIN

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ABSTRACT

Twelve dredging projects are underway or will begin soon. These include both natural and man-made lakes, with lake size and sediment removal up to 205 hectares and 1,720,250 cubic meters, respectively. Solids content of the sediments ranges from 70-80 to 1-5 percent. The projects were designed using a mixture of onsite data collection, predictive models, and professional judgement.

Sediment disposal has limited project implementation, with arsenic being a special problem. Theoretically, sediment concentrations below 4 μg/g could still produce unacceptable contamination of groundwater at the disposal site. One project is being held up, pending completion of laboratory testing.

Organic sediments from Lilly Lake were deposited in an inactive gravel pit and within diked areas on agricultural land. Passage through a spray irrigation system proved impractical. Rapid infiltration of water into the bottom and sides of the settling basins was short-lived due to the self-sealing characteristics of these sediments. Studies are now underway to determine the effect of lake sediment application to upland soils on corn production.

INTRODUCTION

The Office of Inland Lake Renewal (OILR) was created within the Department of Natural Resources in 1974, to protect and rehabilitiate Wisconsin's inland lakes. Water quality had been declining on many lakes and there was general concern for the future well-being of all lakes. There are nearly 15,000 lakes in the State, with a combined area of over 400,000 hectares. They form the foundation of the tourism/recreation economy, the third largest industry. Citizen demand for better environmental protection of these lakes

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prompted the State Legislature to authorize the OILR program through Chapter 33, Wisconsin Statutes. Lake protection/rehabilitation projects are generated and progress through local initiative with technical and financial support from the OILR. Additional assistance is provided by the University of Wisconsin-Extension.

At present, there are 120 projects in some stage of development--data collection/planning/implementation. Rehabilitation projects have utilized various techniques such as aeration, dredging, drawdown, storm sewer diversion, inlake aluminum sulfate treatment, aquatic macrophyte harvesting, improved animal manure handling, streambank erosion control, and several upland conservation methods. However, dredging has been the most often used technique. This paper will describe: 1) the dredging program now underway, 2) the design methodology in usage, 3) the arsenic problem, and 4) the Lilly Lake project.

TEXT

Dredging Program

There are now 12 dredging projects planned or underway in the program (Table 1). Most of these lakes were originally created by dam construction. Two are natural lake basins, although the water level in Little Muskego Lake was raised an additional 2.5 meters in 1938. The lakes range from 4 to 205 hectares. The amount of sediment removal varies from 26,760 to 1,720,250 cubic meters. Hydraulic equipment will normally be used, but in a few cases drawdown has permitted usage of dryland excavation techniques.

TABLE 1. OFFICE OF INLAND LAKE RENEWAL DREDGING PROJECTS

Name	Lake Size (ha)	Watershed Size (km^2)	Impoundment or Lake	Sediment Removal (m^3)
Marinuka	40	404	I	420,500
Upper Willow	96	422	I	150,600
Perch	13	329	I	191,100
Angelo	21	313	I	152,900
Emery	14	28	I	109,300
Martha	5	88	I	38,200
Chilton	4	52	I	26,800
Bugle	14	285	I	160,600
Henry	18	466	I	152,900
Decorah	42	1,425	I	229,400
Little Muskego	205	30	L	1,720,250
Lilly	37	1.6	L	665,200

The sediment characteristics have been diverse. In some cases the materials are dense, primarily inorganic sand, with a solids content of 70-80 percent. This is the usual situation when the lake is located on a major river system. At the other extreme, natural lake sediments are primarily lightweight and organic. The solids content may be as low as 1-5 percent. Chemical composition has also been variable, dependent on previous lake and watershed usage. Analyses are typically performed for at least nitrogen, phosphorus, select heavy metals, and arsenic. Disposal sites have included: 1) lakeside settling lagoons, 2) an inactive gravel pit, 3) diked areas on a golf course and on agricultural lands, and 4) landspreading on cropland.

Design Methodology

There have been two primary objectives in most projects: 1) deepen the lake for improved recreational usage, and 2) control the aquatic macrophyte growth. Before embarking on any inlake dredging project, certain types of information are necessary. Not all of the following are collected on each project; however, there must be a reasonable assurance of environmental improvement and permanency.

1. Radiometric dating. Sediment cores are taken at 1 to 5 sites in the lake basin. These are segmented into depth intervals and analyzed for cesium 137 and/or lead 210 content (Lerman, ed.; 1978). Results have been useful in determining the present and historical rate of infilling, the relative source of the sediment (e.g., streambank versus upland), and the need for improved land usage in the watershed.

2. Sediment delivery. The watershed is examined for streambank and upland erosion. Channel erosion sites are identified by stream survey. The soil loss from each site is quantified on a per year basis, and 50 percent is assumed to enter and be retained in the lake. Soil loss from the uplands is determined by application of the Universal Soils Loss Equation (Wischmeier and Smith, 1965). A number of quarter sections in the watershed are randomly selected for collection of the necessary parameters (Figure 1). The results are then extrapolated to the entire watershed. The actual soil yield to the lake is dependent on the watershed size (Figure 2).

3. Sediment retention. The quantity of sediment trapped in the lake each year is determined by usage of Figure 3. In addition to sediment delivery, it is necessary to determine the annual inflow and the lake's storage capacity in order to use the curve.

   This information, coupled with professional judgment, is used to determine the life span of the lake under existing conditions, the life expectancy of a particular dredging plan, and the need for erosion control in the watershed. In terms of achieving various inlake benefits, several additional considerations are necessary.

4. Habitat preservation. Areas with existing high value for fish and/or wildlife are identified and eliminated from the dredging

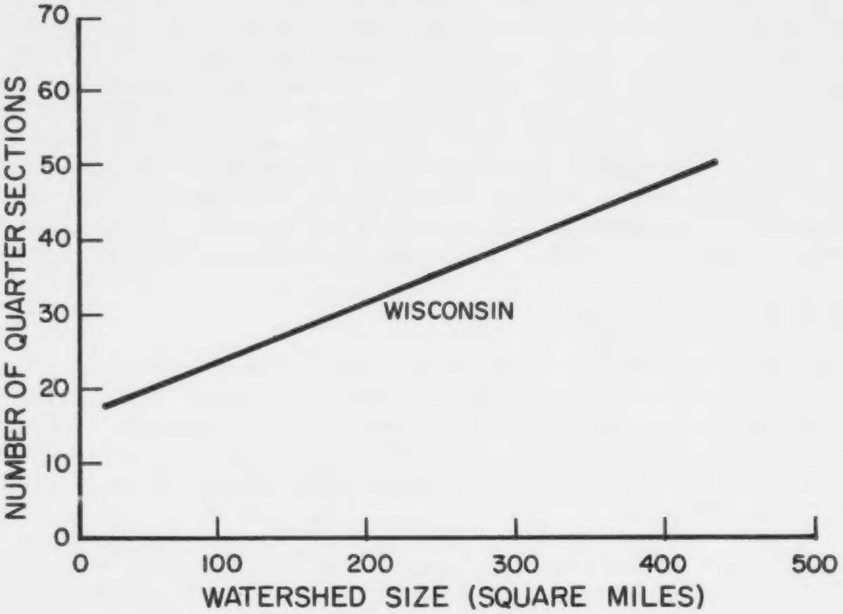

Figure 1. Sample size curve for land use and land treatment (SCS, 1974).

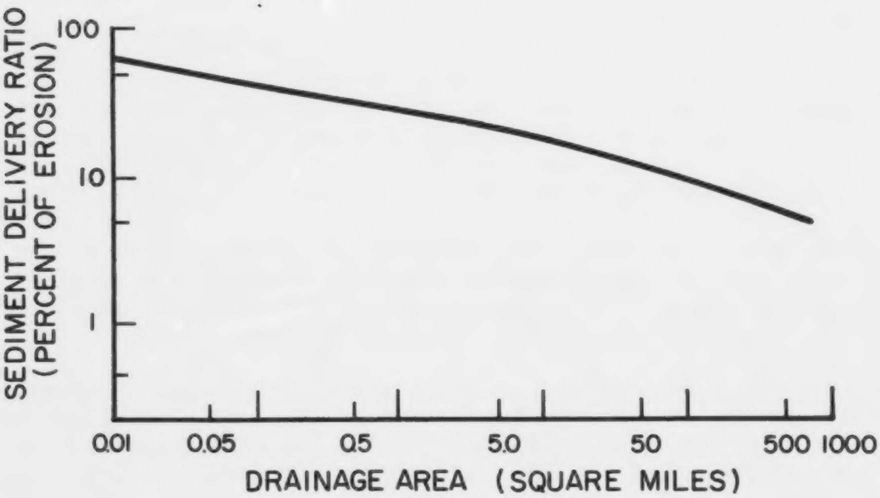

Figure 2. Sediment delivery ratio vs. size of drainage area (SCS, 1971).

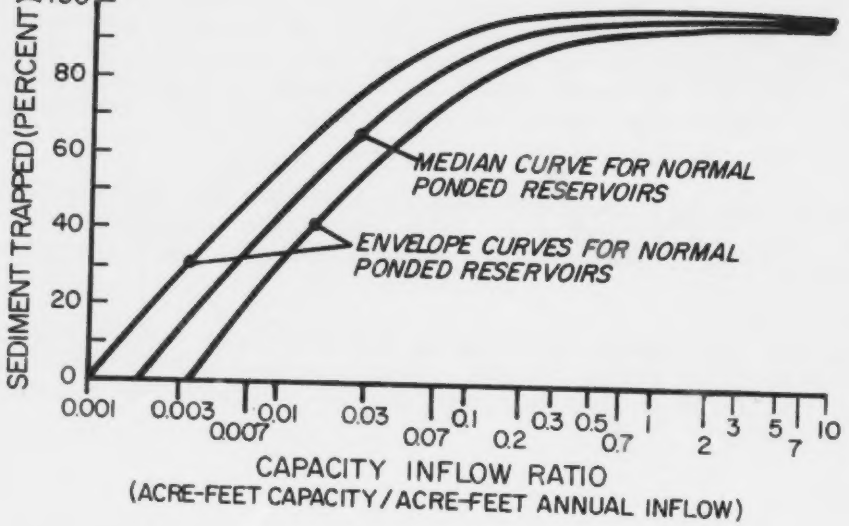

Figure 3. Reservoir trap efficiency as a function of the capacity-inflow ratio (Brune, 1953).

plan. These might be spawning or nursery areas for fish, feeding areas for waterfowl, etc. Emergent and floating leaf plant communities are usually included in this category.

5. Depth of soft sediment. The lake bottom is normally probed to determine the depth of soft sediment. This information is useful in establishing the maximum bottom slope permissible after dredging. If hard bottom (e.g., sand/gravel) is exposed by sediment removal, the slope may be 4 to 1---horizontal to vertical distance. Lightweight, organic sediments represent the other end of the range of possibilities at 10 to 1.

6. Algae/macrophyte control. In projects where macrophyte control is a major objective, the average summer water clarity is determined through periodic measurement by Secchi disc. The maximum depth of weed growth is then estimated by:

$$Y = 0.83 + 1.22X$$

where Y = maximum depth of growth in meters

and X = average summer water clarity in meters (derived from Belonger, 1969 and Modlen, 1970) and the dredging project is designed to produce greater lake depth, thereby preventing growth in that area.

One concern is the potential increase in planktonic algae. It is theoretically possible to produce excessive algae through dredging. Macrophyte control will result in greater availability of incoming nutrients for algal growth. Also, according to Vollenweider (as described in Uttormark and Hutchins, 1978) and Sakamoto (1966) an increase in hydraulic residence time will promote higher algal densities even though the level of nutrient loading remains constant.

Wherever possible, nutrient control measures are applied in the watershed, but when the watershed to lake size ratio greatly exceeds 10 to 1, significant nutrient reductions are generally unattainable (Uttormark et al., 1974).

These are the primary information and predictive tools utilized at present in the decision-making process. They provide the technical framework and financial justification for designing and implementing (or not implementing) dredging projects.

Arsenic

The major impediment to dredging--other than cost--has been the availability of disposal sites. This problem is especially acute for lakes with contaminated sediments. Arsenic is proving to be the single most troublesome element, at least from the standpoint of a potential health hazard.

Prior to 1970, application of sodium arsenite was an acceptable method of controlling the growth of macrophytes. Between 1950 and 1970 about one million kilograms were added to the waters of 167 lakes (Lueschow, 1972). One lake received 142,000 kilograms during that period. Concentrations of arsenic up to 659 μg/g dry weight have since been measured in the surface sediments (Kobayashi and Lee, 1978). Some of these lakes have recently entered the OILR program. In most cases it has been possible to find sites with appropriate characteristics to allow sediment disposal. However, one lake is located in a populous area and the only potential disposal sites are in close proximity to numerous private wells used for drinking water supply. Groundwater contamination is a serious concern.

According to Anderson (1979), arsenic can exist in several different oxidation states and in inorganic or organic configuration. Transformations occur readily, dependent on oxidation potential and pH. Based on theoretical computations, it was determined that even though the average arsenic concentration would be only 4 μg/g in the disposal site, significant groundwater pollution was possible if the element was in a mobile form. Therefore, a three-tiered series of experiments was established to allow prediction of mobility under anaerobesis in the disposal site.

First, sediments were collected from the lake. The concentration of arsenic was determined per species in both the sediments and pore waters. These tests are now underway (Anderson, personal communication). The second level of testing will involve thorough mixing of the sediments with terrestrial soils under aquaeous, anoxic conditions. This will be performed with representative sandy, silty, and clayey soils from each disposal area. If the results at this point are still questionable, undisturbed cores will be taken from a number of sites in the disposal area. Under anoxic conditions in the laboratory, lake sediments will be placed above the core. Measurements will determine the degree of arsenic mobility through the core. Experimentation can be terminated at any point in this process if findings are favorable. If the results do not allay the present environmental concerns, it will be necessary to line the disposal sites with a layer of clay.

Details of arsenic transformation and laboratory experimentation are available from Prof. Anderson. On this project as well as others involving potentially hazardous materials in the sediments, long-term monitoring plans will be established at the disposal sites.

Lilly Lake Project

Lilly Lake is located in southeastern Wisconsin. The drainage basin is 155 hectares in size, and there are no surface inlets or outlets. The lake covers 37 hectares and has a mean depth of 1.4 meters. Maximum depth was only 1.8 meters with greater than 10.7 meters of underlying organic sediments (e.g., 62 percent). The water content of the sediments ranged from 90-98 percent (Table 2). The bottom was covered by dense rooted macrophytic growth. Inlake production and deposition was causing an infilling rate of 0.5 centimeter per year (e.g., radiometric dating using the PB 210 method).

TABLE 2. SOLIDS CONTENT OF THE SEDIMENT (September 20, 1977; 4 locations)

Depth Into Sediments	Percent Dry Solids	Percent Water
1.5 meters (5 ft.)	2.4 - 8.6	96.4 - 97.6
3.7 meters (12 ft.)	3.1 - 4.3	95.7 - 96.9
6.1 meters (20 ft.)	4.8 - 9.4	90.6 - 95.2

The dredging operation was designed to remove about 600,000 cubic meters of sediment, increasing the maximum depth to 6.1 meters. Dredging was initiated in July, 1978 and continued until November. It commenced again in May, 1979 and was completed by September. During 1978, a hydraulic dredge was used to pump the sediments through a 30 centimeter polyethylene pipe almost 3 kilometers to a settling basin. In 1979 the sediment was also applied to 15 hectares of agricultural land. The effect of dredging on the lake and disposal sites are being monitored through a grant from the U.S. EPA. Investigations began in 1976, and will continue into 1982. Selective findings relating to sediment disposal are presented herein.

The settling basin was originally an inactive gravel pit cut into a hillside resulting in slopes approximately 10 meters high on one side. Top soil had been removed from the 18 hectare area. Two earthen dikes were constructed--one across the open side of the pit and another through the middle of the area, thereby creating 2 sub-basins. The dikes and bottom of the basins were composed primarily of clay, whereas the side slopes were mostly sand/gravel/rocky material. Nine observation wells, including three piezometer nests, were established around the area to monitor for changes in water level. In addition, nine domestic wells were periodically sampled for water quality.

It was initially anticipated that the materials would be removed from the lake without requiring entrainment of carriage waters. Therefore, the capacity of the site was designed to hold sediments only, with eventual drying primarily through evaporation. However, because the inlake sediments did not flow as expected--vertical walls were reportedly created by dredging--and the dredge was relatively slow-moving, the pumped slurry consisted of 55 percent carriage water in 1978. As the mixture was deposited in the basin, the sediments settled to the bottom and were overlain with carriage water. At first this water was able to infiltrate rapidly into the highly porous side slopes.

The influence of water infiltration was greatest near the basin and decreased with distance away (Figure 4). At 18 meters the water level eventually exhibited a 3 meter increase versus wells further away. At 75 and 150 meters the change was less than at 350 meters, suggesting that the effect of infiltration became negligible somewhere between 18 and 75 meters.

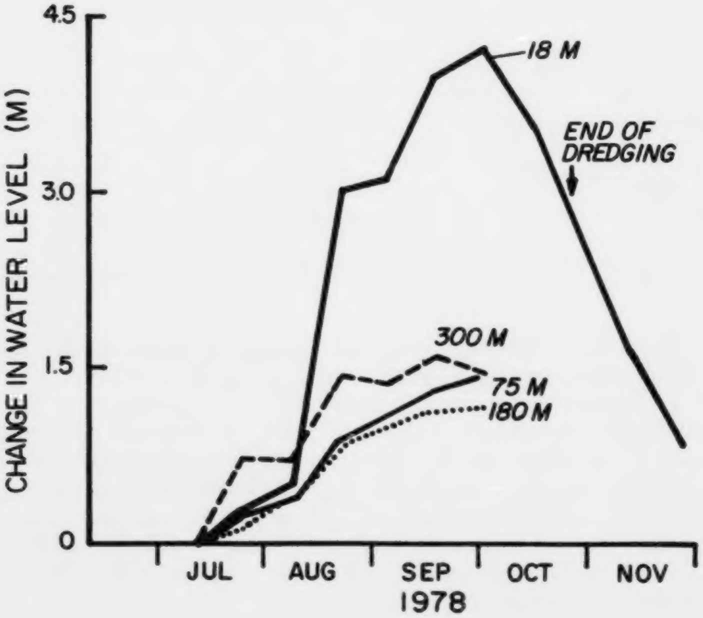

Figure 4. Change in water levels at various distances from the settling basin.

About one month before dredging ended in 1978, the water level at 18 meters began to recede, dropping over 3 meters by late November. The basin was essentially filled from mid-October through November and the change in sediment/water elevation was slight. Apparently the basin's side slopes were sealed soon after the sediment/water reached a particular elevation. Rapid infiltration was possible only during the filling process. In 1978 the water levels in the observation wells were not significantly influenced by the disposal operation and dropped in response to dry climatic conditions despite the near-full settling basin.

Although substances with long-term toxicity had not been used historically in the lake or watershed, there initially was some concern, especially regarding nitrogen transport away from the disposal area. However, examination of the sediment pore waters at the disposal site revealed no problem, nor is it likely that one will develop in the future (Table 3). The water quality of the domestic wells has not changed to date. These wells range in distance from the settling basin - 75 to 360 meters - and in depth to the well point - 10 to 90 meters. In addition, a review of the borings and well logs in the area suggests that all of the wells may be hydraulically separated by clay layers from any materials put into the settling basin.

TABLE 3. COMPOSITION OF THE SEDIMENT AND PORE WATERS

Parameter*	Sediments (μg/g)	Pore Water (μg/l)
Aluminum	219,400	250
Barium	3,040	Less than 400
Iron	136,000	3,200
Copper	270	3
Zinc	3,790	40
Chromium	540	Less than 3
Phosphorus	6,600	1,300
Ammonia-nitrogen	--	9,700
Nitrite/nitrate-nitrogen	--	Less than 20
Total nitrogen	27,000	--

* pH of 7-8

In 1979 agricultural land was also used for sediment disposal. Dikes were erected on 15 hectares of land, creating 6 individual basins capable of holding up to 2 meters of material. Spray irrigation equipment was additionally installed on 10 hectares. Unfortunately the fibrous organic sediments consistently clogged the nozzles (5 centimeter orifice) within 30 minutes operation. The technique should be useful for disposal of carriage waters from settling basins in future projects; however, dispersal of the sediment/water slurry will be infeasible without major system modifications.

Six observation wells were installed around the perimeter of the 15 hectare site and monitored periodically for changes in water level and quality. The well borings revealed an extreme variability in the stratigraphy of the area, with some borings passing through only sand/gravel while others went through clay. Nevertheless, following a temporary rise the water levels dropped throughout the summer, and little change has been detected in water quality. Apparently the organic nature of the sediment causes it to form a nearly-impermeable seal over the bottom and sides of the settling basins, thereby greatly inhibiting infiltration into, and water quality effects on, the groundwater system.

After sufficient drying has taken place, next spring the dikes will be removed; the organic sediments will be plowed into the terrestrial soils; and the area will be returned to agricultural production. Planned studies on this area will determine residual effects on soil chemical and physical properties including nitrate accumulation in the soil profile. The following will be investigated concurrently: 1) nutrient mineralization rate for sediments from 10 diverse lake types; 2) effect of sediments on corn production in greenhouse experiments (4 sediment types selected on the basis of their ranges in C/N and N/P ratios will each be added to a sand and a silt loam soil at 4 different application rates); and 3) field tests using corn will be performed with a representative silt loam soil from the area and the Lilly Lake sediments at 4 rates of application.

Preliminary work has been promising (Corey and Peterson, personal communication). These studies are particularly important to the OILR program, because if lake sediments significantly increase crop production and the benefits have been clearly demonstrated, agricultural lands will become more available for future dredging projects. Also, a market might ultimately be developed for the sediments, thereby reducing the total cost for a project.

ACKNOWLEDGEMENTS

Financial support for the Lilly Lake investigation is being provided through a grant from the U.S. Environmental Protection Agency's research laboratory in Corvallis Oregon; Spencer Peterson, project officer. Rick Beauheim (hydrogeologist) assisted in analyses of monitoring data from the Lilly Lake sediment disposal sites.

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RELEASE OF PHOSPHORUS FROM LAKE SEDIMENTS

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ABSTRACT

The relationship between the content of various forms of phosphorus in lake sediments and the amount of phosphorus released under aerobic and anaerobic conditions was studied. Total phosphorus content in the sediment of Lake Kasumigaura was highest at the 0-5 cm surface layer and decreased with depth. The constant value below 15 cm was consistent with the decrease of iron-bound phosphorus content (Fe-P). The amount of phosphorus released from the sediments was proportional to the decrease of Fe-P under both aerobic and anaerobic conditions. Under anaerobic conditions, 90% of the Fe-P initially held in the sediments was released in 55 days. Using dialysis apparatus, maximum growth yield of algae was shown to be linearly dependent on the amount of phosphorus released under aerobic conditions.

INTRODUCTION

Bottom sediment is known to play an important role in the eutrophication process. Particularly in a lake restoration program, we cannot ignore the effects of the sediments on the phosphorus budget. In restoration programs that have been completed, there are examples where water quality did not improve to the desired levels because of phosphorus release from the sediments (1).

Many studies have reported on the rates of phosphorus release from the sediment. These studies were conducted under defined environmental conditions, aerobic and/or anaerobic, using specific lake sediments (2-4). The results, however, differ greatly from case to case, even among sediment samples from the same lake, depending on the experimental conditions and/or characteristics of the sediment. Thus, because of this lack of basic knowledge, it is doubtful if we can estimate the real rate of phosphorus release.

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The purpose of this study was to clarify the relationship between the amount of released phosphorus and the amount of various fractions of phosphorus contained in the sediment. Phosphorus in the sediments was fractionated before and after phosphorus release experiments and the fractions which contribute the phosphorus release were determined.

MATERIALS AND METHODS

Sediment Sampling

Sediment samples were taken with a coring device to minimize sample disturbance. Immediately after sampling, the core was divided at 2- to 5-cm intervals and taken to the laboratory. Samples for phosphorus analyses were freeze-dried; those for phosphous release experiments were refrigerated at 5°C.

Phosphorus Analysis of the Sediments

Total phosphorus content (T-P) was determined by the ignition method of Anderson (5,6). To determine total inorganic phosphorus (I-P), it was extracted by 1 N HCl (or 1 N H_2SO_4) for 16 hours at room temperature and then the concentration of orthophosphate in the extract was analyzed. Total organic phosphorus (O-P) was determined by subtracting the value of I-P from that of T-P (7).

Several procedures to fractionate I-P in sediments and/or soils have been reported (8-10). Many are based on the procedure proposed by Chang and Jackson (11) as is the procedure used in this study (Figure 1). Three fractions of phosphorus extracted by NH_4-F, NaOH, and H_2SO_4 were defined as aluminum-bound phosphorus (Al-P), iron-bound phosphorus (Fe-P), and calcium-bound phosphorus (Ca-P), respectively. Phosphorus concentrations extracted by these methods were determined by the Ascorbic Acid Reduction Method (12) as a concentration of PO_4-P.

Phosphorus Release Experiment

The sediments were suspended homogeneously in a lake water medium and the rate of phosphorus release was measured under both aerobic and anaerobic conditions. The apparatus shown in Figure 2 was designed to study the effect of aerobic sediments on algal growth (13). This apparatus is composed of two L-type vessels with a volume of approximately 300 ml. A Millipore membrane filter (pore diameter = 1.2 microns) between the vessels is supported by porous glass discs.

Phosphorus-free AAP (algal assay procedure) medium (14) was used in this study. The working volume was 250 ml for both vessels. Sediment was suspended (sediment vessel) in one vessel and *Selenastrum capricornutum* (green alga) was located in the other (algal vessel). Both vessels were stoppered with cotton plugs, and mounted on a rotary shaker illuminated by fluorescent light at 25 ± 2°C. The light intensity at the medium surface was about 4,000 lux. In this system, the only phosphorus supply to the algae was that contained in the sediment and it had to pass through the membrane filter.

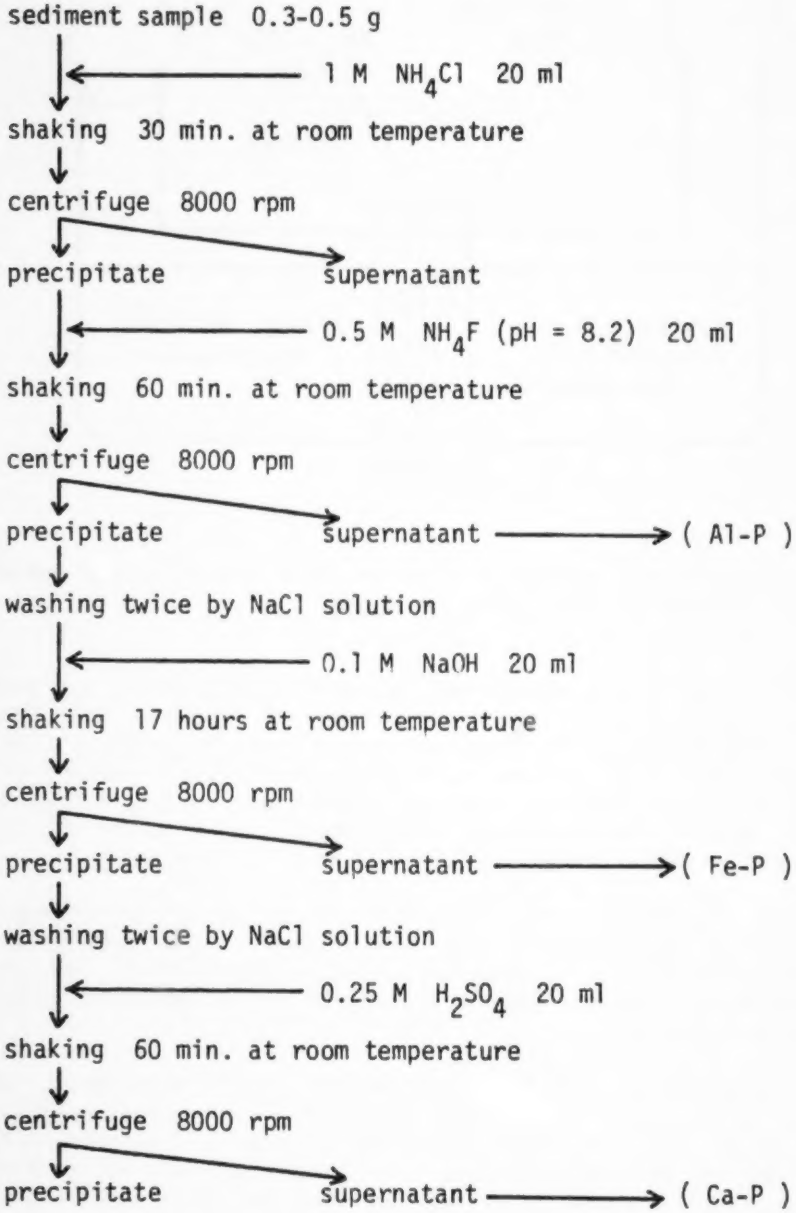
sediment sample 0.3-0.5 g
1 M NH_4Cl 20 ml
shaking 30 min. at room temperature
centrifuge 8000 rpm
precipitate
supernatant
0.5 M NH_4F (pH = 8.2) 20 ml
shaking 60 min. at room temperature
centrifuge 8000 rpm
precipitate
supernatant
(Al-P)
washing twice by NaCl solution
0.1 M NaOH 20 ml
shaking 17 hours at room temperature
centrifuge 8000 rpm
precipitate
supernatant
(Fe-P)
washing twice by NaCl solution
0.25 M H_2SO_4 20 ml
shaking 60 min. at room temperature
centrifuge 8000 rpm
precipitate
supernatant
(Ca-P)

Figure 1. Inorganic phosphorus fractionation in lake sediments.

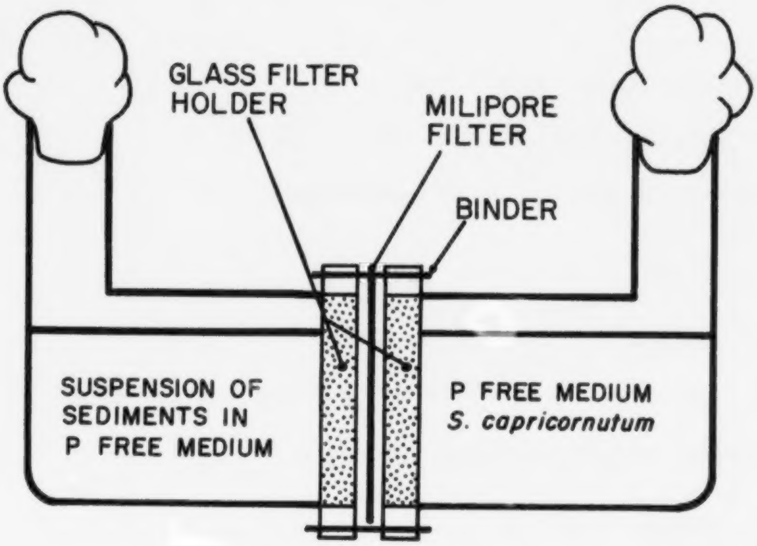

Figure 2. A dialysis apparatus used to study the effect of aerobic phosphorus release on the growth of algae.

Algal biomass was monitored by cell counting and by mean cell volume measurement with a Coulter Counter, Model ZF equipped with Mean Cell Volume Computer (Coulter Electronics, Inc., Hialeah, Florida). The data on cell number and mean cell volume were converted to dry weight of algal cells, referring to a calibration prepared separately, i.e., a sample of the algal suspension was filtered through a Millipore filter (Type HA, pore diameter = 0.45 microns), and dried at 90°C for 24 hours before weighing.

At the stationary phase of algal growth, total phosphorus concentration in the medium was determined by the Ascorbic Acid Reduction Methods after digesting cells with potassium persulfate. Both before and after incubation of the sediments, the phosphorus fractions in the sediments were determined as described previously.

The phosphorus release experiments under anaerobic conditions were conducted as follows. A sediment sample of 1 g (as dry weight) was suspended in lake water which had been filtered through a Millipore filter and poured into a 1,000-ml Erlenmeyer flask. This sediment-lake water mixture was mixed homogeneously by a magnetic stirrer and the anaerobic condition was maintained by passing nitrogen gas through at 20 ± 2°C. The PO_4-P concentration in the water phase was determined by sampling the sediment-water mixture. The suspended sediments were removed by centrifugation followed by filtering supernatant through a Millipore filter. Phosphorus fractions in aerobic and anaerobic sediment were determined by using the same procedure (Chang and Jackson fractionation followed by Ascorbic Acid Reduction Method).

RESULTS AND DISCUSSION

Fractionation of Sediment Phosphorus

Figures 3 and 4 show the vertical distribution of phosphorus fractions in the sediment core samples from Lake Kasumigaura. As shown in Figure 3, T-P decreased with depth and showed a constant value below 8 cm. The ratio of I-P to T-P was 70% at the surface layer and decreased with depth. Thus the amount of T-P decrease seems consistent with the decrease of I-P. Similar results were obtained with the other sediment samples from Lake Kasumigaura, Lake Hinuma (eutrophic), and also from Lake Shoji-ko (eutrophic). These results correspond to those reported by Wildung *et al*. (5) where they concluded that most of the O-P was highly resistant to microbial activity. Thus, when phosphorus release from sediments is to be studied, detailed analyses of I-P are required rather than T-P or O-P.

Figure 4 shows the vertical distribution of phosphorus fractions in the sediment core samples from the center of Lake Kasumigaura. T-P decreased from 1.06 mg P/g in the 0-5 cm surface layer to 0.78 mg P/g in the 10-15 cm layer. Below 15 cm, T-P content was consistent around 0.7 mg P/g. I-P in this case was the sum of Al-P, Fe-P, and Ca-P, and was almost the same as that determined by 1 N HCl extraction (15). Although Al-P did not change significantly with depth, Fe-P and Ca-P decreased with depth but were relatively constant below 15 cm. The highest values for all three fractions were observed at the 0-5 cm surface layer. The most notable decreases in phosphorus content of sediments with depth were those for the Fe-P fraction from the surface to 15 cm.

Phosphorus Release Under Aerobic Conditions

Sediment samples for this study came from Lake Kasumigaura (Tsuchiurairi Bay, depth = 2.5 m). Only the surface layer of the core, from 0 to 3 cm, was used. Figure 5 shows the relationship between the maximum growth yield of *S. capricornutum* and the amount of total phosphorus released from the containerized sediment and diffused into the algal vessel. Both the maximum growth yield and the amount of phosphorus released increased linearly with the increase in the amount of suspended sediment. Dissolved oxygen concentration in the sediment vessel was around 7 mg/l throughout the experiment.

Table 1 shows the variations of phosphorus fractions in the sediment before and after aerobic incubation. The amount of sediment suspended in this case was 0.72 g per 250 ml of medium. T-P of 0.18 mg P/g decreased from 1.14 mg P/g to 0.96 mg P/g by the release of phosphorus. Among three fractions of I-P, Fe-P of 0.17 mg P/g decreased from 0.3 to 0.13 mg P/g, whereas neither Al-P nor Ca-P decreased. Thus the amount of T-P decrease appears to correspond with that of Fe-P.

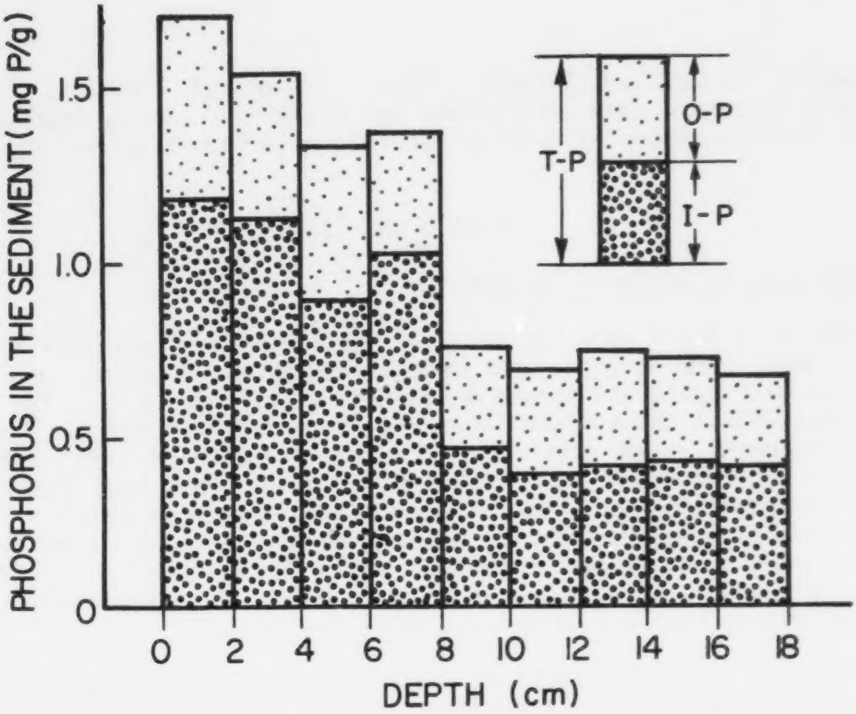

Figure 3. Vertical distribution of phosphorus content in the sediment of Lake Kasumigaura (11/25/77).

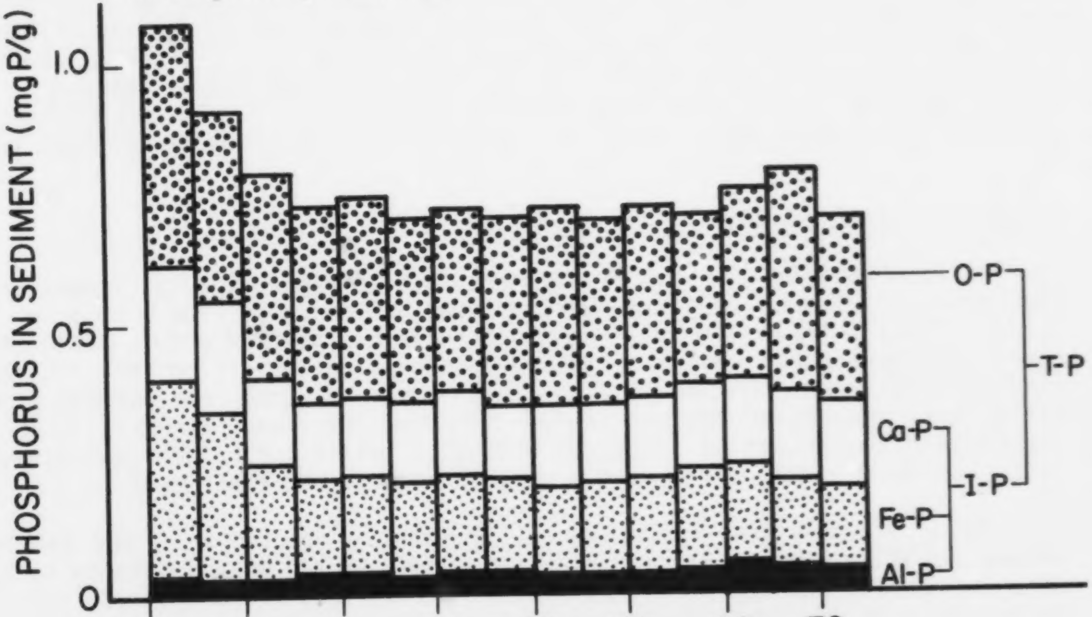

Figure 4. Vertical distribution of phosphorus fractions in the sediment of Lake Kasumigaura.

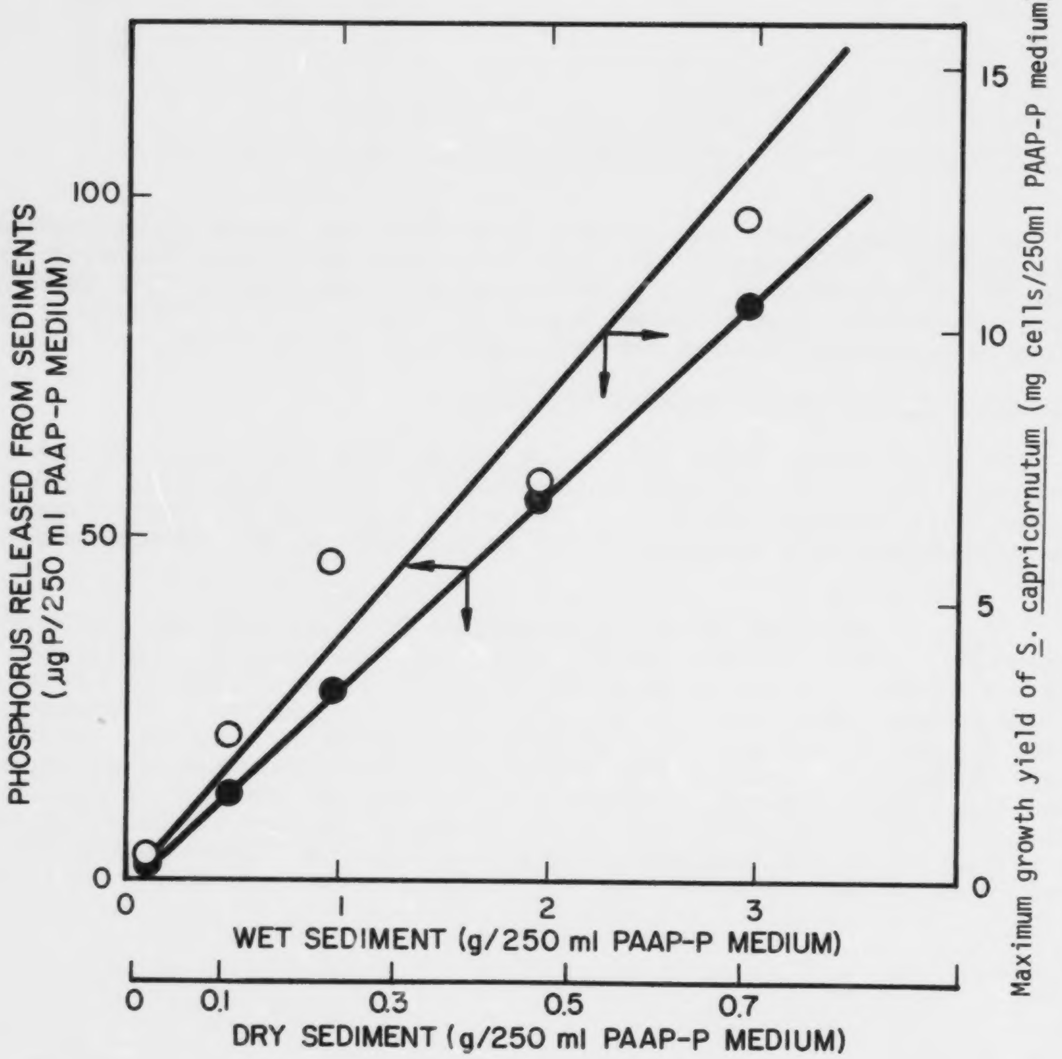

Figure 5. Maximum growth yield of S. capricornutum and the amount of phosphorus released under aerobic conditions in the algal vessel as affected by the amount of the sediments of Lake Kasumigaura suspended in the sediment vessel.

TABLE 1. VARIATION OF PHOSPHORUS FRACTIONS IN THE SEDIMENT OF LAKE KASUMIGAURA UNDER AEROBIC CONDITIONS

Phosphorus Fraction	Initial Content mg P/g	Content After Incubation mg P/g	Difference mg P/g
Total Phosphorus	1.14	0.96	-0.18
Inorganic-P	0.67	0.50	-0.17
Al-P	0.11	0.10	-0.01
Fe-P	0.30	0.13	-0.17
Ca-P	0.26	0.27	+0.01
Organic-P	0.47	0.46	-0.01

In the cases where the sediment taken from Lake Towada (oligotrophic), Lake Shikotsu (oligotrophic), and Lake Yunoko (eutrophic) were incubated, the same result was obtained, i.e., T-P decrease corresponded to Fe-P decrease (data not shown). Figure 6 shows a positive linear relationship between the amount of phosphorus released and Fe-P fraction in the sediments.

PHOSPHORUS RELEASE UNDER ANAEROBIC CONDITIONS

Sediment samples for this study came from two sites, Akanoi-wan (polluted, depth = 1.8 m) and Kusatsuyamada-oki (unpolluted, depth = 3.0 m), in the South Basin of Lake Biwa. The surface layer and the deeper layer of the sediment core were suspended in the surface water at the sediment sampling point.

Figure 7 shows the release of phosphorus into lake water under anaerobic conditions. After the 25th day of incubation, the phosphorus concentration did not change. Oxidation-reduction potential during incubation was maintained between -58 mV and -123 mV. Table 2 shows the variation of phosphorus fractions in the sediments before and after anaerobic incubation. Although the Al-P and Ca-P fractions from Akanoi-wan sediment increased after incubation for unknown reasons, a remarkable decrease of the Fe-P fraction was observed. The relationship between the amount of phosphorus released and the Fe-P fraction in the sediments is shown in Figure 8. Approximately 90% of Fe-P fraction was released from the sediment under anaerobic conditions.

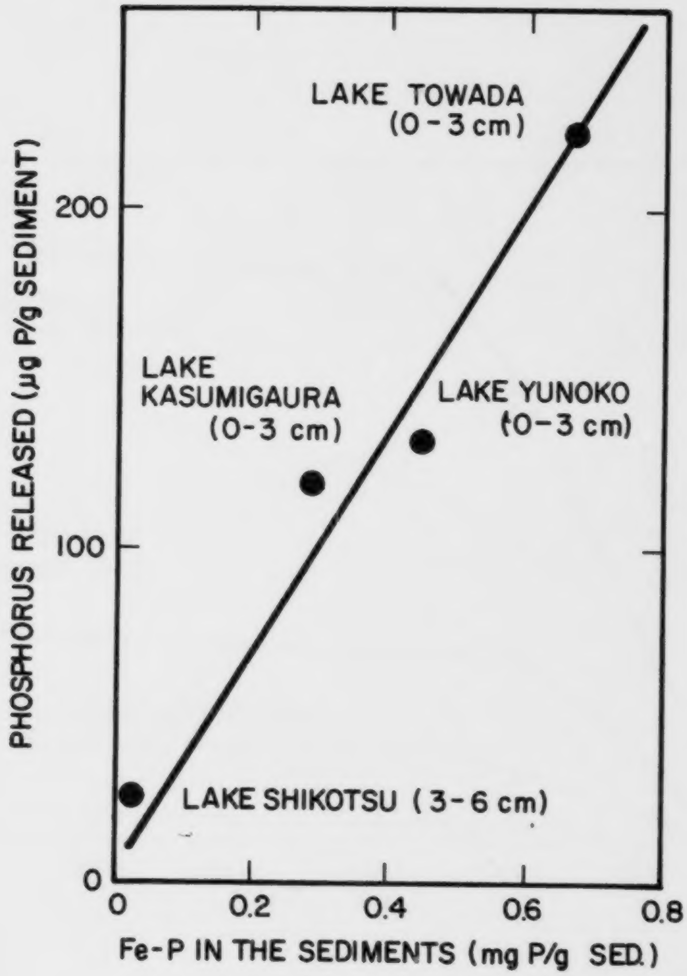

Figure 6. A relationship between Fe-P fraction in the lake sediment and the amount of phosphorus released under aerobic conditions.

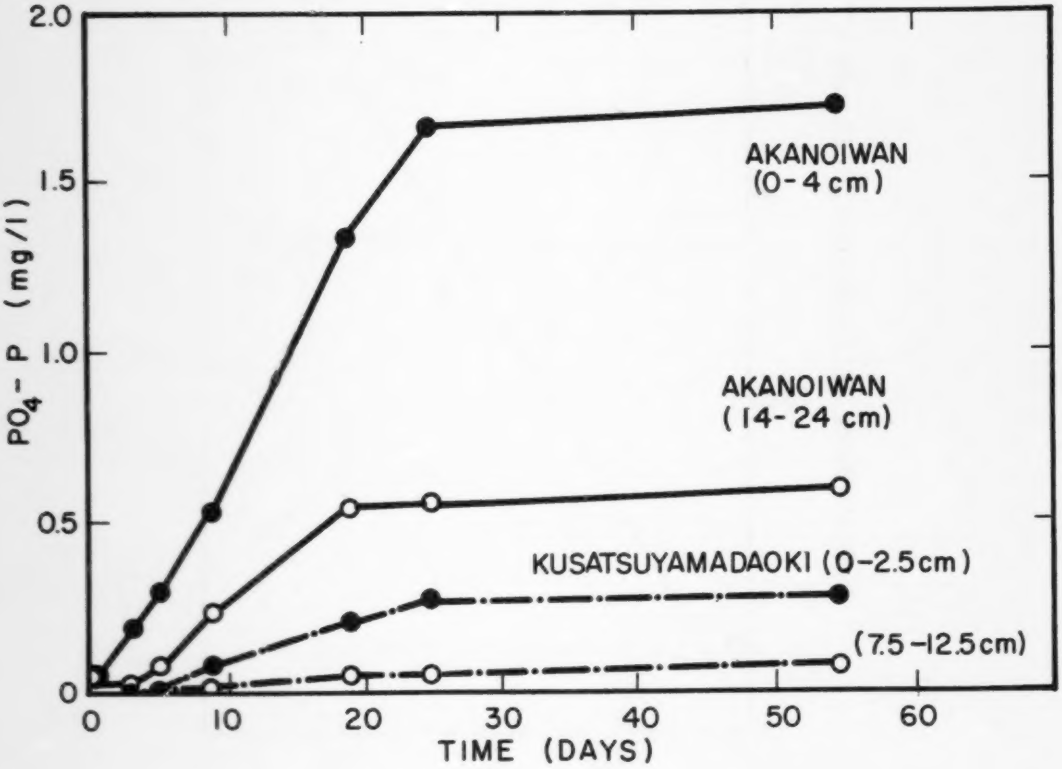

Figure 7. Release of phosphorus (as phosphate) during the incubation of the sediments of Lake Biwa under anaerobic conditions.

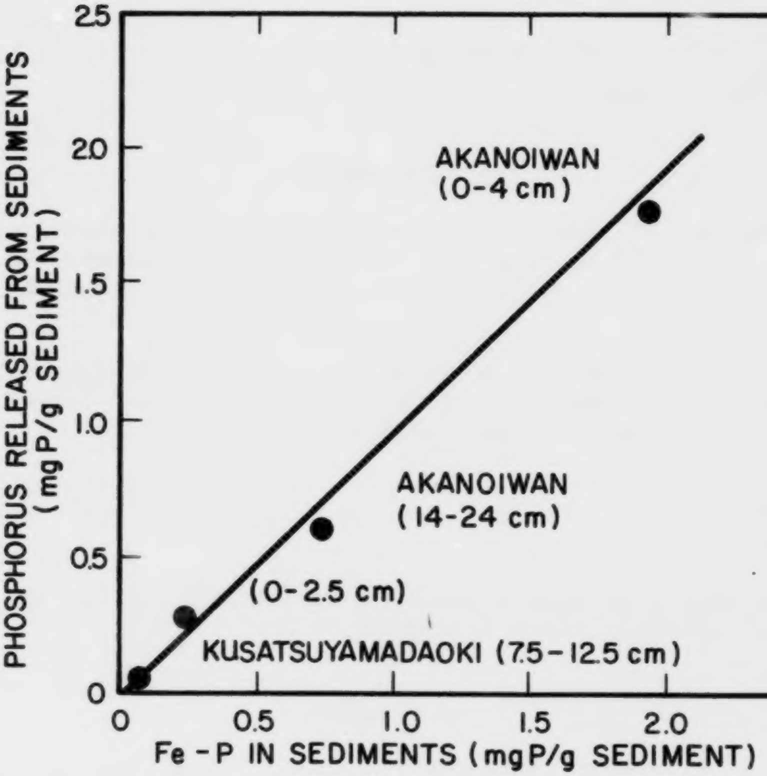

Figure 8. Relationship between Fe-P fraction in the sediment and the amount of phosphorus released under anaerobic conditions.

TABLE 2. VARIATION OF PHOSPHORUS FRACTIONS IN THE SEDIMENT OF LAKE BIWA UNDER ANAEROBIC CONDITIONS

Sample	T-P mg P/g	Al-P mg P/g	Fe-P mg P/g	Ca-P mg P/g
Akanoi-wan (0-4 cm)	2.70	0.09 0.47*	1.98 0.42*	0.28 0.43*
Akanoi-wan (14-24 cm)	1.11	0.02 0.14*	0.75 0.07*	0.12 0.20*
Kusatsuyamada-oki (0-2.5 cm)	0.72	0.04 0.05*	0.25 0.04*	0.20 0.18*
Kusatsuyamada-oki (7.5-12.5 cm)	0.47	0.03 0.03*	0.08 0.02*	0.08 0.08*

* Contents after incubation.

CONCLUSIONS

Fractionation of sediment phosphorus and the measurement of phosphorus release were conducted to establish the relationship between the distribution of phosphorus fractions and the amount of phosphorus released. The results were as follows:

(1) T-P in the sediment of Lake Kasumigaura was highest at the surface layer, decreased with depth, and was constant below 15 cm. This variation was consistent with the change of Fe-P.

(2) Both the amount of aerobically released phosphorus and the maximum growth yield of algae were proportional to Fe-P fraction in the sediments.

(3) The amount of anaerobically released phosphorus was about 90% of the Fe-P fraction in the sediments.

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RELEASE, DISTRIBUTION, AND IMPACTS OF POLYCHLORINATED BIPHENYLS (PCB) INDUCED BY DREDGED MATERIAL DISPOSAL ACTIVITIES AT A DEEPWATER ESTUARINE SITE

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ABSTRACT

The aquatic disposal field investigation initiated in Elliott Bay, Puget Sound, Washington, in February 1976, was designed to evaluate the ecological effects of open-water disposal of dredged material. The experimental disposal site was located at a depth of 60 m in a marine estuary with generally weak circulation. Approximately 114,000 m^3 of dredged material contaminated with polychlorinated biphenyls (PCBs) were dumped at the site from split-hull barges. The material presented in this paper is limited to a discussion of PCB impacts. It includes:

(a) a summary of results from the studies conducted between 1976 and 1977 to assess short-term impacts during and after the disposal of PCB contaminated sediments at the Elliott Bay disposal site;

(b) a presentation of preliminary results from the continuation studies initiated in February 1979 to determine long-term impacts of the disposal operations.

The preliminary data indicate that both dredged-material deposit and the associated PCBs appear to be stable. No major long-term impact on benthic organisms is apparent from this analysis.

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INTRODUCTION

The aquatic disposal field investigation (ADFI) in Elliott Bay, Seattle, Washington, was initiated in February 1976 as part of the Environmental Impacts and Criteria Development Project of the U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi. This study was one of four major research projects within the Dredge Material Research Program designed to evaluate the ecological effects of open-water disposal of dredged material.

Among the four coastal area disposal sites selected for the ADFI (Lake Erie, off Ashtabula, Ohio; the mouth of the Columbia River; Gulf of Mexico off Galveston, Texas; Elliott Bay, Puget Sound, Washington), Elliott Bay was the only deepwater estuarine location where dredged material disposal by barges was investigated and where maintenance dredging of the Duwamish River (discharging into the bay) involved the disposal of sediments highly contaminated with polychlorinated biphenyls (PCBs). Therefore, the potential adverse biological consequences that could result from the release of PCBs induced by these dredging and disposal activities made this site particularly interesting.

The material presented in this paper is limited to a discussion of PCB impacts and includes:

(a) a summary of results from the studies conducted between 1976 and 1977 to assess short-term impacts during and after the disposal of PCB contaminated sediments at the Elliott Bay disposal site;

(b) a presentation of preliminary results from the continuation studies initiated in February 1979 to determine long-term impacts of the disposal operations.

In this paper, only the PCB distributions, sediment physical characterization, and the impacts on benthic macrofauna will be considered. These and other aspects of the short-term studies have been discussed in detail elsewhere (U.S. Army Engineers, 1978).

DESCRIPTION OF THE STUDY AREA

Elliott Bay is situated midway on the eastern shore of the central basin of Puget Sound (Figure 1). The surface area of the bay is approximately 14.4 km^2 and is defined by Magnolia Bluff as its northwest boundary and on the southwest by Duwamish Head. The volume of the bay comprises approximately 1 percent of the volume in the main basin (McClellan, 1954) and 0.5 percent of the total Puget Sound volume. Bottom topography is characterized by steep marginal shore slopes around an internal basin of about 130 m in depth. This basin slopes gently to the northwest until it merges with the central Puget Sound basin.

The southern portion of the bay is divided into two smaller basins by a bottom ridge that slopes northwesterly from the northern end of Harbor Island

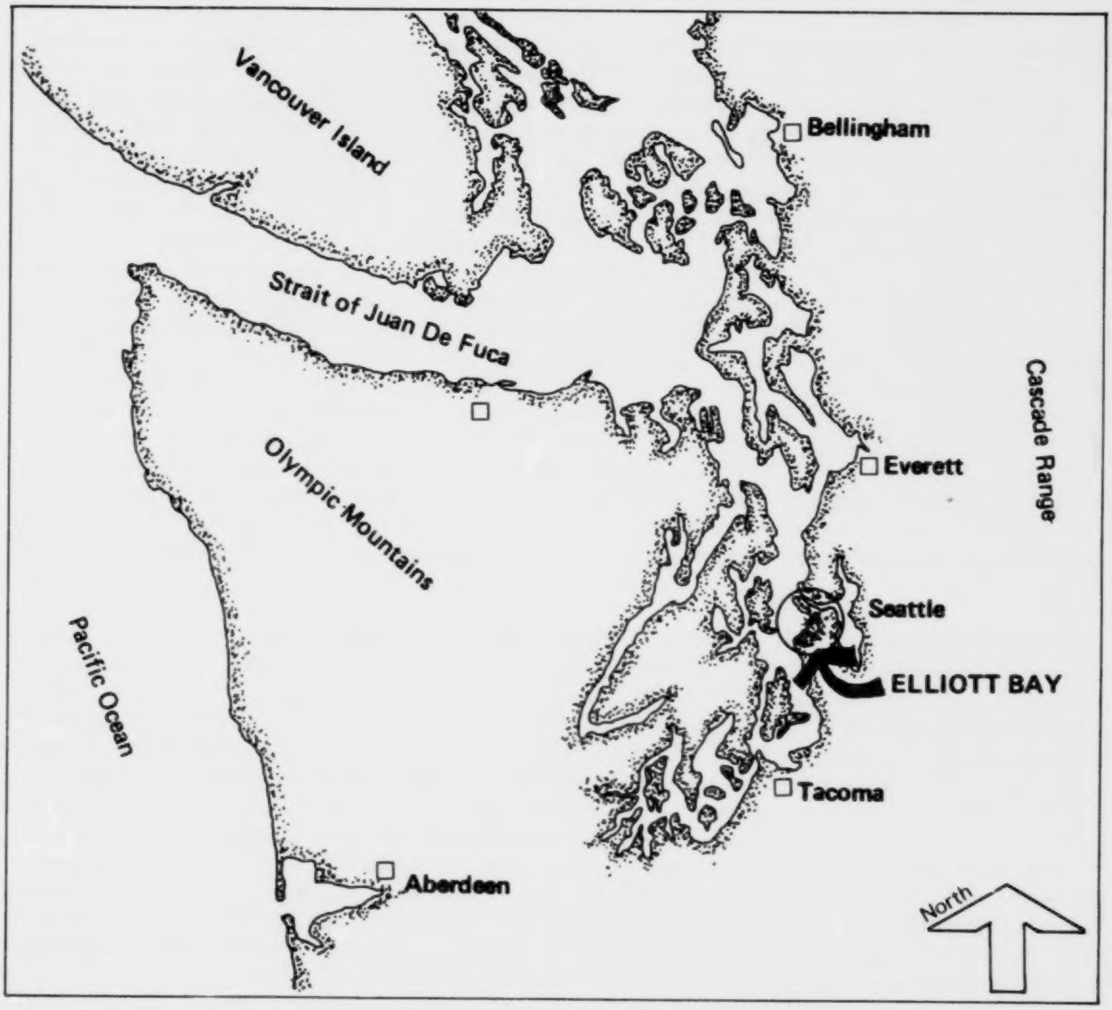


Figure 1. Puget Sound and Elliott Bay

and extends to the center of the bay. This ridge may represent a delta built by the Duwamish River, which discharges into the southern portion of the bay.

The circulation in Elliott Bay is predominantly induced by tides. Tide fluctuations (3.2 m mean tide range) generate a weak, generally counterclockwise flow in the upper layers (<50 m) of the bay, with water from the main basin entering around Duwamish Head. While deepwater exchange between the bay and the main basin has no topographic restrictions, circulation in the deep layers is probably limited except during periods of deep water renewal within the entire Puget Sound system.

The Duwamish River provides freshwater input to Elliott Bay at an average annual rate of about 1,300 cfs (U.S. Environmental Protection Agency, 1974). The flow is highly seasonal, reflecting variations in precipitation and snowmelt. The river discharge normally increases in late fall and again in late spring. The lower Duwamish forms a vertically stratified salt-wedge estuary with net outflow of fresh to brackish water at the surface and net inflow (upriver) of saline Elliott Bay water at depth. The highly variable flow of freshwater is nearly always seaward. However, the instantaneous movement in both layers may be either upstream or downstream. At its mouth, the river is split and discharges into Elliott Bay around both sides of Harbor Island. Dredging of the western channel and a shallow sill at the south end of the eastern channel result in the majority of the water exchange taking place via the West Waterway. The freshwater discharge forms a low salinity surface plume (1-15 m) in the southern portion of the bay. The behavior of this plume reflects a response to both tidal currents and wind stress. In the absence of strong southerly winds, the plume is "compressed" into the southern bay around the river mouth by flood tides. During ebb tides, the plume normally drifts northward, spreading along the northeastern waterfront and following the shoreline until its identity is lost by mixing with Puget Sound surface water. As a result, the influence of the river discharge is felt primarily in the southern and southeastern portions of Elliott Bay and along the Seattle waterfront.

The presence of PCBs in Puget Sound has been known since 1972. In general, PCB concentrations were found to correlate with sites of increased industrial and municipal activity with no apparent temporal trends. The highly industrialized Duwamish Estuary contained the highest PCB concentrations observed in the sound. Elliott Bay, which receives the Duwamish River discharge, also was found to contain elevated PCB levels showing a spatial distribution in surface sediments that decreased with distance from the mouth of the river. A recent examination of PCB levels in the sediments of Elliott Bay and the Duwamish River suggests that the history of PCB input into this area has been sporadic over a fairly long period of time. Sediment cores often show marked differences in both the PCB types and their total concentrations as a function of the core depth. A detailed discussion on these aspects has been presented elsewhere (Pavlou and Dexter, 1979; Hom, 1979).

SAMPLING SCHEME

Short-Term Studies

The dredging and disposal operations were initiated in February 1976 and completed in March 1976. A clamshell bucket dredge and two split-hull barges of approximately 1100 m^3 combined capacity were used in the operation. The total volume of material disposed in Elliott Bay was approximately 114,000 m^3. The source of these sediments was a 1.88 km stretch of the upper Duwamish Estuary between river km 6.3 and 8.2. The disposal site was located over the 60 m depth isoline due north of the mouth of the West Waterway (47°37'41" N; 122°21'42" W). The sixteen-station sampling grid comprised an area of 0.13 km^2, as shown in Figure 2. The two reference sites were also located in 60 m of water and positioned east and west of the disposal site. The west reference site historically has received the least impact from the municipal, commercial, and industrial activities of the Seattle area. Water flow over this location originates primarily from the main basin of Puget Sound rather than from the interior of Elliott Bay. The east reference site has received effluents from the Duwamish River and unknown contributions of contaminants from shipping, nearby shore-based activities, and from a number of sewage overflow discharges along the Seattle waterfront.

During the short-term studies, effects of the disposal operations were examined by monitoring three barge dumping episodes on each of two separate days. Rapid time series of whole water and suspended particulate matter samples for PCB analysis were collected at the center of the disposal grid. Reference values were measured each day before and after the monitoring period at each of the east and west reference areas and at the mouth of the Duwamish River. Samples were collected at the surface (0-1 m), 10 m off the bottom and 1 m off the bottom at each station.

Replicate water and suspended particulate matter samples were collected at intervals of two days, 10 days, one month, three months, six months, and nine months after the cessation of all dumping at two stations near the center of the disposal grid and at the same three reference stations mentioned above. The same sampling depths also were used.

Replicate sediment cores from the river, disposal grid, and east and west reference stations were collected 10 days before initiation of dredging. Additional cores were collected at 20 Elliott Bay stations (Figure 2) during the nine-month post-disposal sampling period discussed above.

Benthic macrofauna were collected in replicate or triplicate using a 0.1 m^2 van Veen grab sampler at the same stations and schedule as was used for the sediment samples. The macrofauna were sieved using 1 mm screens on board ship.

Long-Term Studies

Since the baseline study results indicated that the disposed material extended beyond the boundaries of the original station grid (see below), the spatial coverage for the long-term studies was expanded. In this manner also,

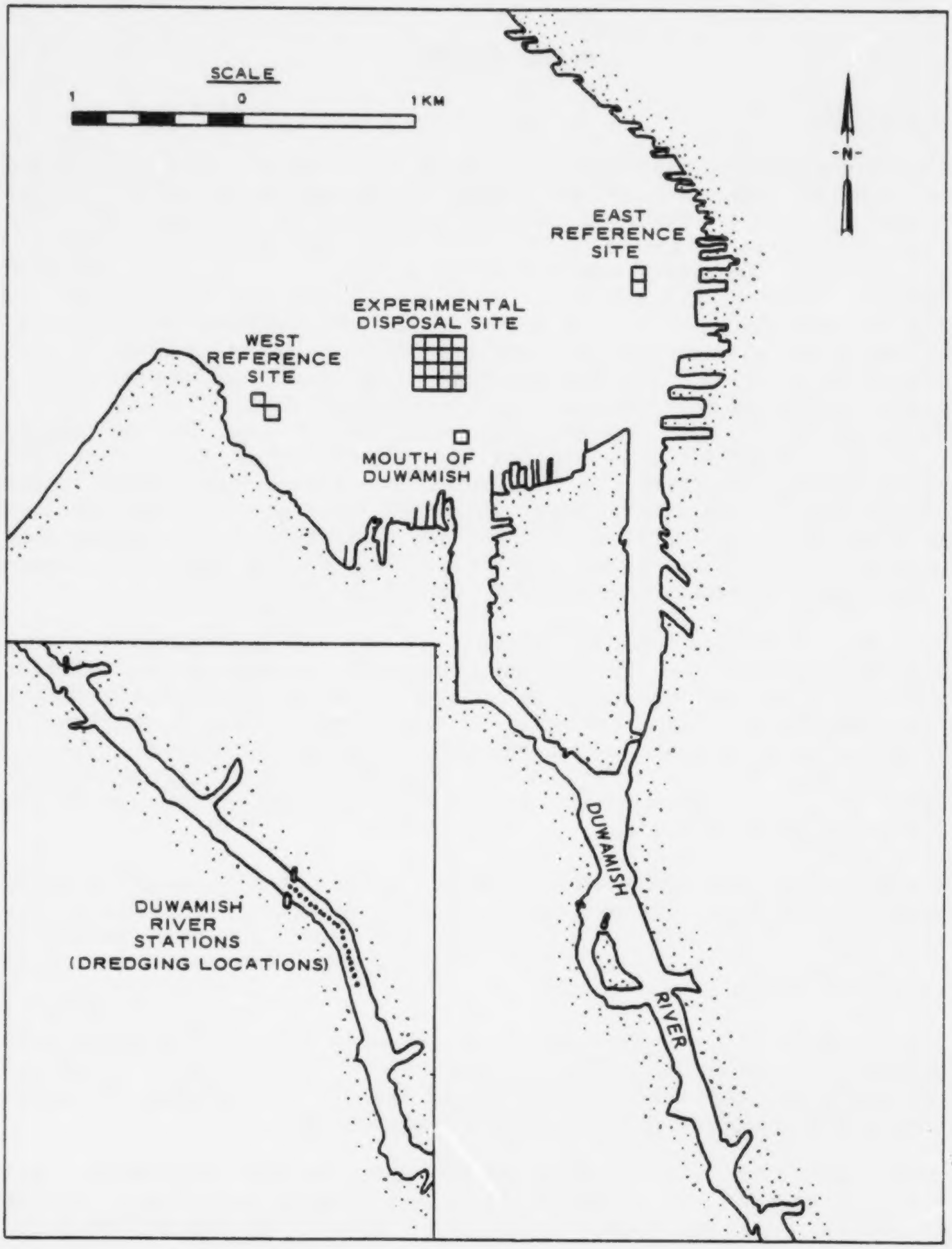


Figure 2. Original Station Grid

a better comparison could be made between impacted and non-impacted zones, specifically with respect to estimating patterns and variability in the surrounding biotopes for evaluating recovery from perturbation. A new sampling scheme was therefore developed and is briefly summarized below.

Of the original grid of sixteen stations in a four-by-four square pattern (Figure 2), the four corner stations were retained; the original side stations, together with the reference stations, were eliminated. Four new side stations within the grid were chosen at intermediate locations. Eighteen stations were chosen outside the grid at points located at 35 m increments of distance from the center of the grid at random angles. For each subsequent cruise, stations outside the grid were re-randomized. A schematic diagram of the station randomization scheme is shown in Figure 3. The new configuration for the first sampling cruise is shown in Figure 4. This method was used to establish an unbiased selection of reference stations, while still maintaining temporal continuity for some of the original grid stations.

Essentially the same sampling schemes were employed in the long-term studies, with the major exception that the sediment cores were split into a greater number of horizons (up to 5), usually at visual or textural discontinuities.

All analytical procedures used during these studies have been described in detail elsewhere (U.S. Army Engineers, 1978).

SPATIAL CONFIGURATION OF THE DISPOSAL MOUND

Detailed bathymetric maps of the disposal areas are shown in Figures 5 and 6, with contour intervals of one foot and five feet, respectively. These maps were generated from data collected in December 1978 by the U.S. Army Engineers, Seattle District, but they are essentially the same as the previous post-disposal surveys.

The most prominent feature is the dredged material "mound" of approximately 3 meters depth defined by the 190-foot contour near the center of the grid. Detailed comparisons of pre- and post-disposal surveys indicated that dredged material is present throughout most of the grid area, predominantly in the northern section.

Only minor changes were detected when comparisons were made between the previous post-disposal bathymetric surveys and the more recent data. These changes are illustrated in Figure 7, which compares bottom-depth profiles along east-west transects through the grid center and at 100 foot intervals north and south of the centerline.

STABILITY OF THE DREDGED MATERIAL DEPOSIT

In order to determine the physical stability of the disposal mound, measurements of current velocity, salinity, temperature, transmissivity, and pressure were taken. In May 1979 one array of Aanderaa current meters and the

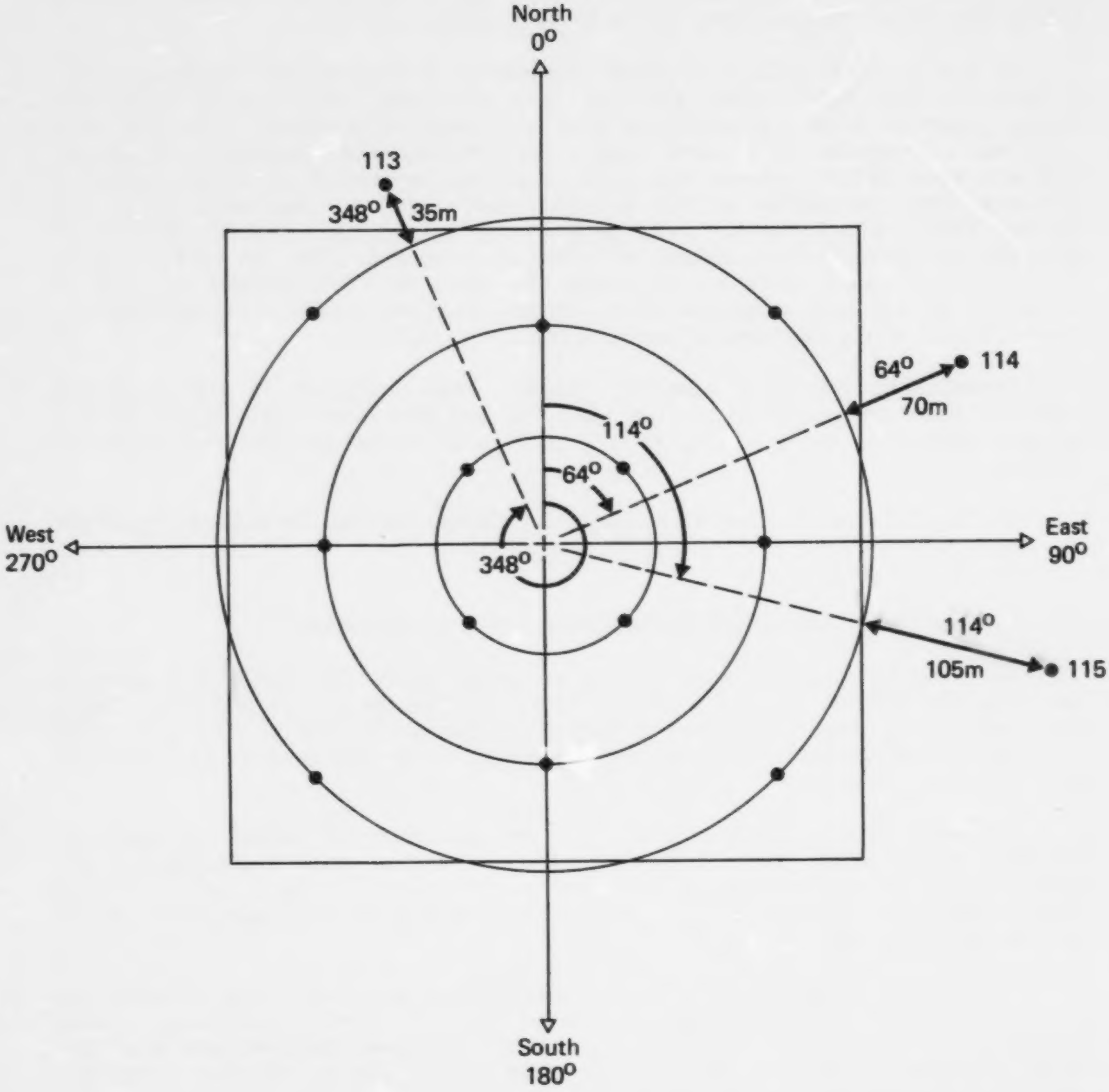


Figure 3. Station Location Randomization Procedure

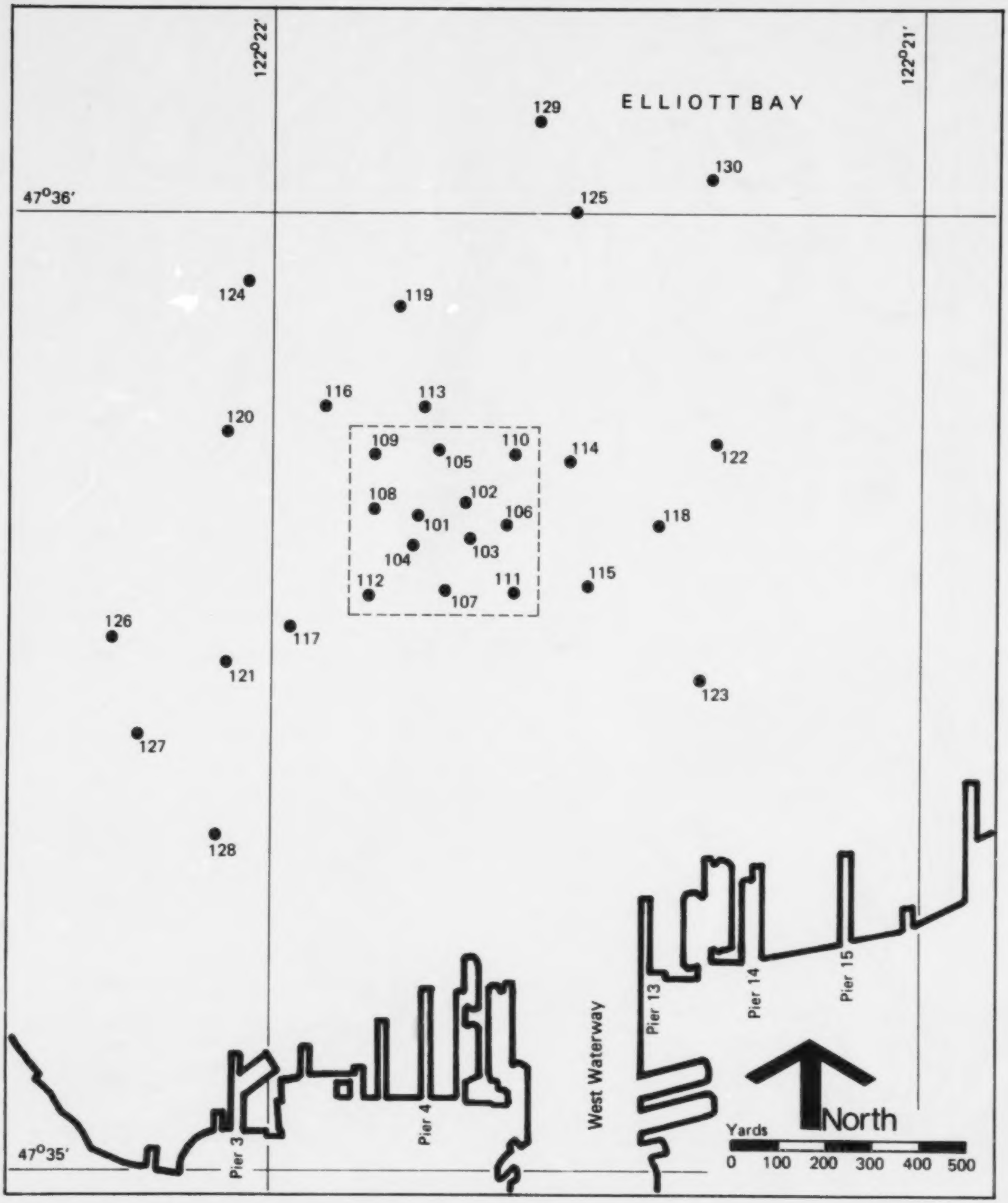


Figure 4. Station Locations - Reconnaisance Cruise February 1979

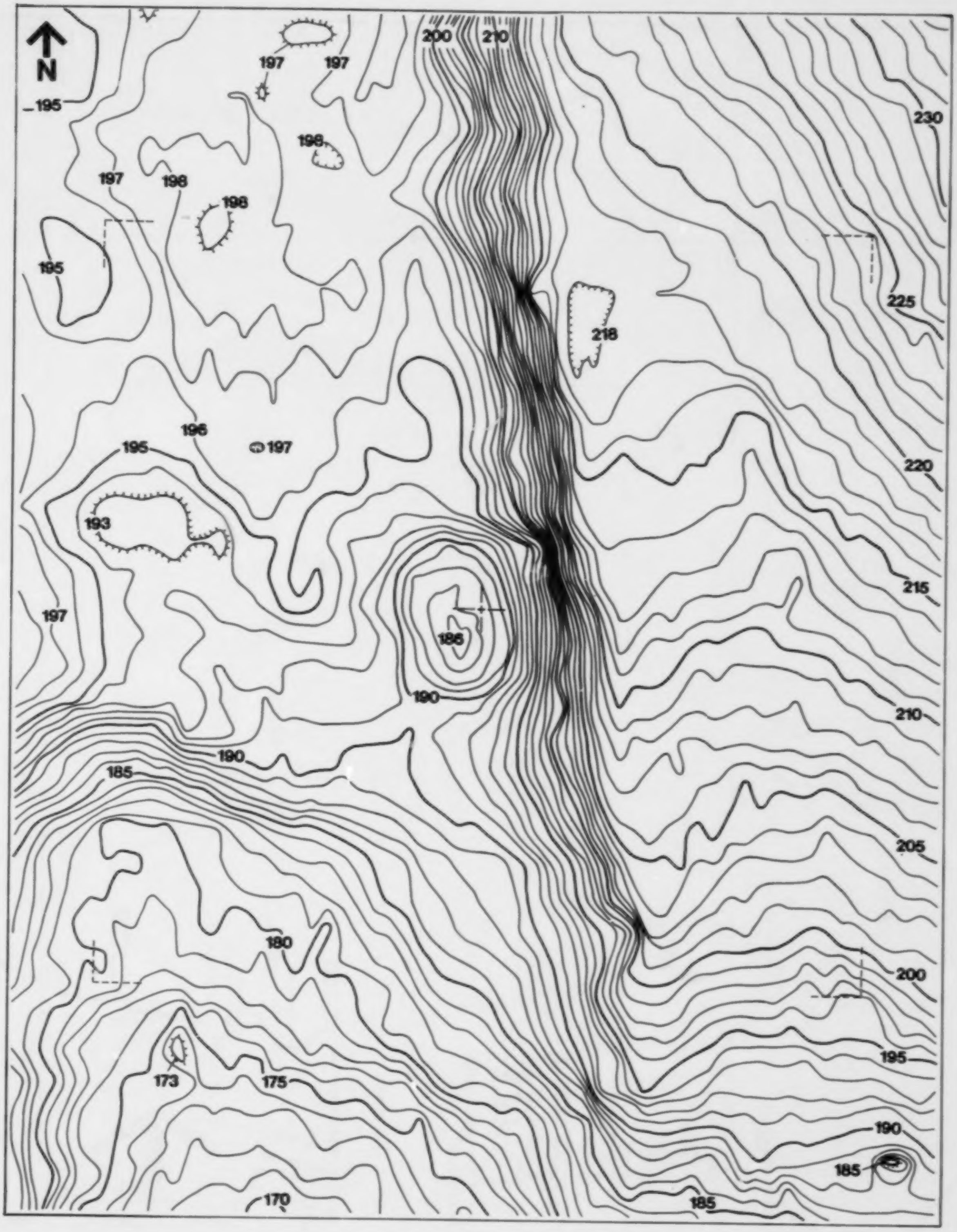


Figure 5. Bathymetric Map of Disposal Site (1 foot contours)
Dashed Lines Define the Area of the Original Sampling Grid

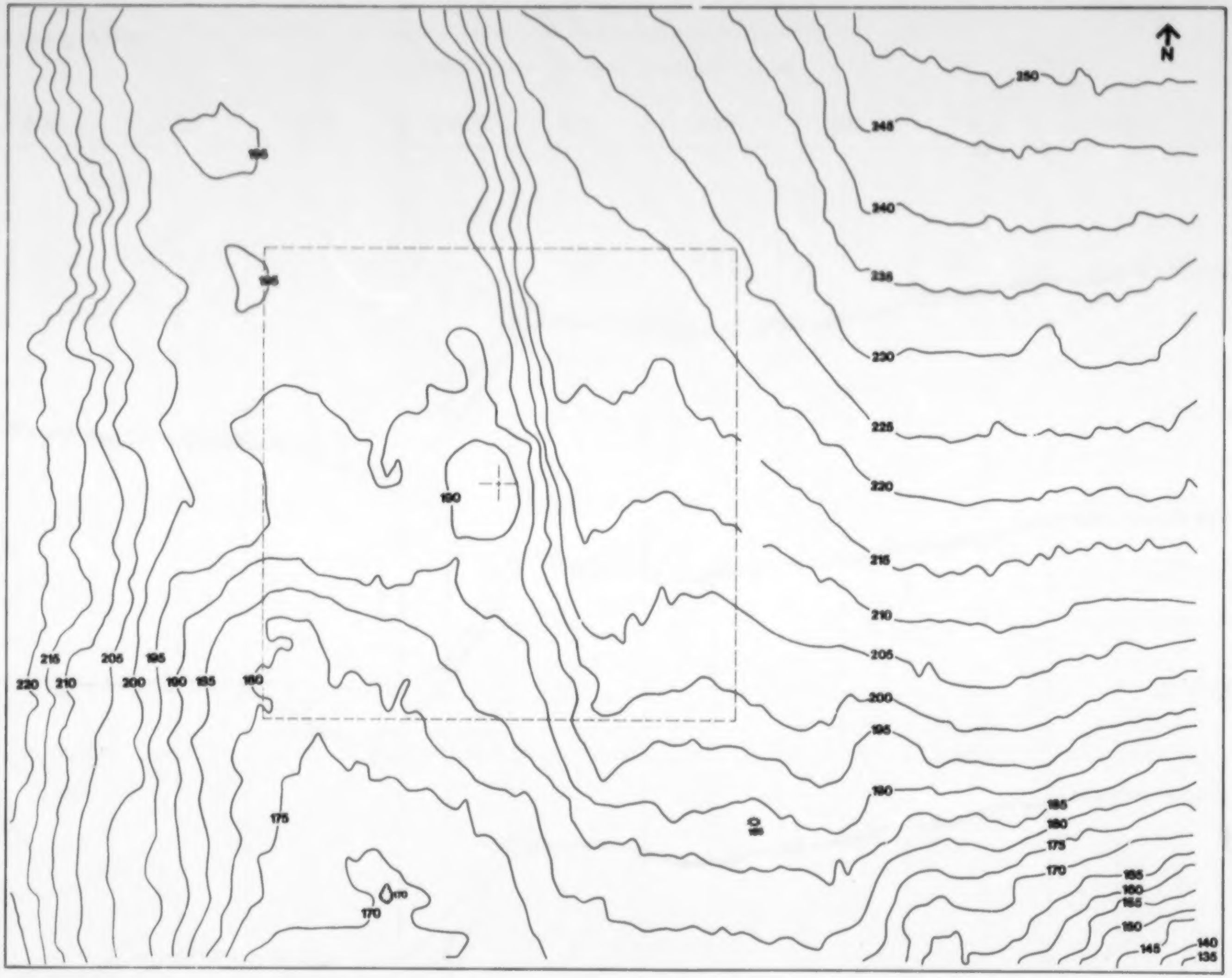


Figure 6. Bathymetric Map of Disposal Area (5 foot contours)
Dashed Lines Define the Area of the Original Sampling Grid

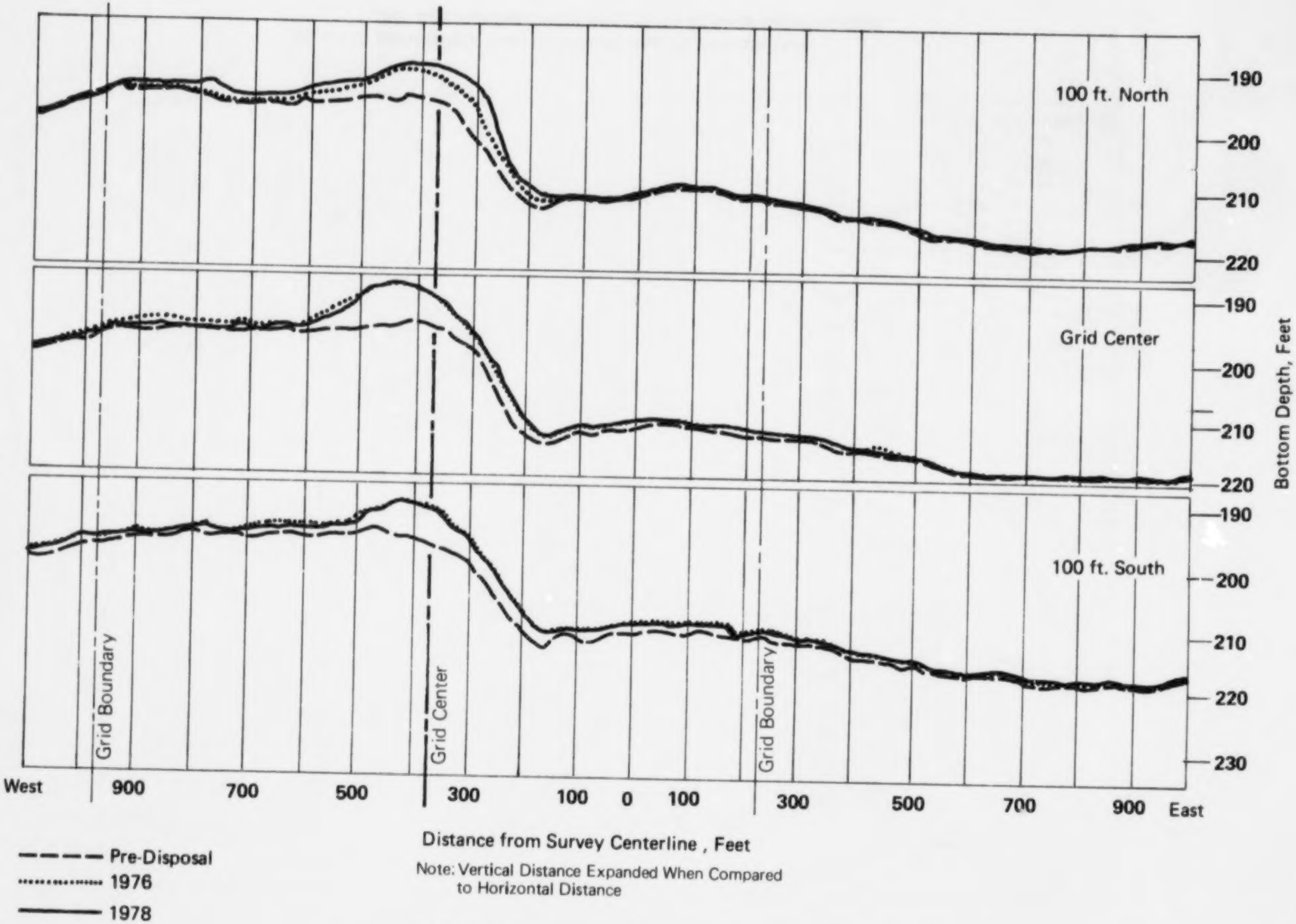


Figure 7. East-West Transects Through the Disposal Site

Sediment Dynamics Sphere (SDS) tripod system, from the Department of Oceanography at the University of Washington, were deployed for 40 days in the study area. The current meter array and tripod were placed about 30 m apart in a relatively flat area just north of the original grid. The data for the SDS system and current meter records are now being processed and will include tables of velocity components, current speed and direction, temperature, conductivity, pressure and transmissivity measured at 15 minute intervals during the deployment.

Sediment Transport Calculations

Based on the data obtained, some preliminary sediment transport calculations have been made. The critical boundary shear stress (T_c) is dependent on grain size and can be estimated for a range of phi sizes. A critical shear velocity ($u_{\star c}$) was calculated for each T_c from the relation

$$T_c = \rho u_{\star c}{}^2$$

A range of critical Reynolds numbers ($R_{\star c}$) was then calculated from

$$R_{\star c} = \frac{u_{\star c} D}{v}$$

where D = grain size diameter and v = water viscosity.

Using Nikuradze's diagram a Z_o was determined for each $R_{\star c}$.

The Karman-Prandle equation

$$u = \frac{u_\star}{k} \ln \frac{Z}{Z_o}$$

relates the velocity u, at depth Z above the bottom to the shear velocity $u_\star$ and the natural log of $\frac{Z}{Z_o}$. Knowing the critical values of $u_\star$ and Z_o, a critical value of u can be calculated. If the actual values of u exceed the critical value, sediment may be moved.

The results of these calculations are summarized in Table 1. It should be noted that these calculations are based on cohesionless sediments, whereas the sediments in the study area are cohesive. Therefore, the values calculated (Table 1) for the critical velocities (u_c) which would be required to move the various sediment classes probably represent minimum velocities.

As can be seen in Table 1, current speeds actually observed at 2 m above the bottom exceeded 15 cm/sec 0.5 percent of the time during the deployment period. Currents exceeded 20 cm/sec for a total of 0.25 percent of the time. The maximum velocity (one 15 minute reading) during the 40-day deployment was 23.3 cm/sec. Velocities were predominantly 10-12 cm/sec and appeared to change direction in response to the tides.

TABLE 1. SEDIMENT TRANSPORT CALCULATIONS

phi Size	D(cm)	u_{*c} (cm/sec)	Z_o (cm)	u_c (cm/sec at 2 m)
2	2.5×10^{-2}	1.43	1.0×10^{-3}	43.6
3	1.25×10^{-2}	1.33	1.1×10^{-3}	40.3
4	6.25×10^{-3}	0.96	1.5×10^{-3}	28.3
5	3.12×10^{-3}	0.81	1.8×10^{-3}	23.5
6	1.56×10^{-3}	0.71	2.0×10^{-3}	20.4
7	7.81×10^{-4}	0.62	2.3×10^{-3}	17.6

Observed Values

currents exceeded 15 cm/sec : 0.5% of deployment period
currents exceeded 20 cm/sec : 0.25% of deployment period
maximum value: 23.3 cm/sec.

Based on these calculations, sediments of phi size 7 or greater could have been resuspended a maximum of 0.5 percent of the time during the deployment. Sediments of phi size 6 could have moved a maximum of 0.25 percent of the time. Sediments of 5 phi or less should not have moved. In comparison with the observed sediment texture (see below), these calculations suggest that some fine sediments could have moved during a small percentage of the deployment period. However, the effect of the sediment cohesion probably prevented significant resuspension. This conclusion is supported by transmissometer readings which indicated that the suspended sediment concentrations never exceeded 1 mg/l. No significant peaks were recorded.

Sediment Texture Analysis

Sediments were analyzed for percent water, percent organic matter and grain size distribution. Standard sieve and pipette techniques were used to measure grain size. Sands were measured in 1/4 phi intervals, silts in 1/2 phi intervals and clays in 1 phi intervals. The relationship between phi ($\emptyset$) size and grain diameter is given by $\emptyset = -\log_2 D$ where D = grain diameter in millimeters. The basic statistical analysis included calculations of the percents of gravel, sand, silt and clay; sand-to-mud ratio; sorting; skewness; Kurtosis; and mean and median phi. The following discussion is applicable to samples collected during the reconnaissance cruise (February 1979). Analysis of all samples collected in May 1979 is not yet complete.

There was significant variability in grain size between some samples taken at the same station. For example, differences of 36 percent in sand content among three replicate surface samples were measured. This variability must be kept in mind when reviewing the general results presented below.

Plots of percent sand and mean phi size samples for the February data are presented in Figures 8 and 9, respectively. Where more than one replicate was analyzed per station, the average percent sand and average mean phi were used. A north-south orientation is apparent in the percent sand distribution. There was a middle band of coarse sediment bordered by less coarse bands. Most of the surface sediments are between 30 percent and 60 percent sand. A somewhat similar pattern is seen in the mean phi size distribution although it is not as distinct as in percent sand. Most samples had mean phi in the silt size class.

All of the samples analyzed had large standard deviations indicating poor sorting. As a result, it is difficult to characterize the sediments since the distributions of various parameters tend to produce different spatial patterns.

Part of the shipboard processing of the gravity cores included recording a physical description of each core. This description included overall length, color and texture variations, odor, layering, and any other notable characteristics. In and near the grid area, a layer of black sediment underlain and/or overlain by greenish gray sediment was observed in many samples. Considering the location of the samples and the known characteristics of the dredge material, this black layer was most probably dredge material. In outlying areas cores were frequently fairly uniform with depth and were generally greenish-gray or grayish-brown in color. Since the usual characteristics noted in the cores corresponded poorly with the usual sediment textured paramters, patterns in individual phi sizes were examined. The phi size classes were ranked for each sample in order of abundance. Similarities in the six most abundant phi sizes were apparent and the majority of the sediments could be classified into five subjectively defined sediment groups based on these recurring patterns. The criteria defining each sediment type are presented in Table 2.

TABLE 2. DEFINITION OF SEDIMENT TYPES

Type	Criteria
a	4.5ϕ and 3.25ϕ in 6 most abundant, without 2.75ϕ
b	4.5ϕ, 3.25ϕ and 2.75ϕ in 5 most abundant
c	4.5ϕ not present in 6 most abundant
f	12ϕ most abundant, 4.5ϕ 2nd, no 9ϕ
g	12ϕ most abundant, 4.5ϕ 2nd, and 9ϕ 3rd-5th most abundant
Misc.	Does not fit any other sediment type.

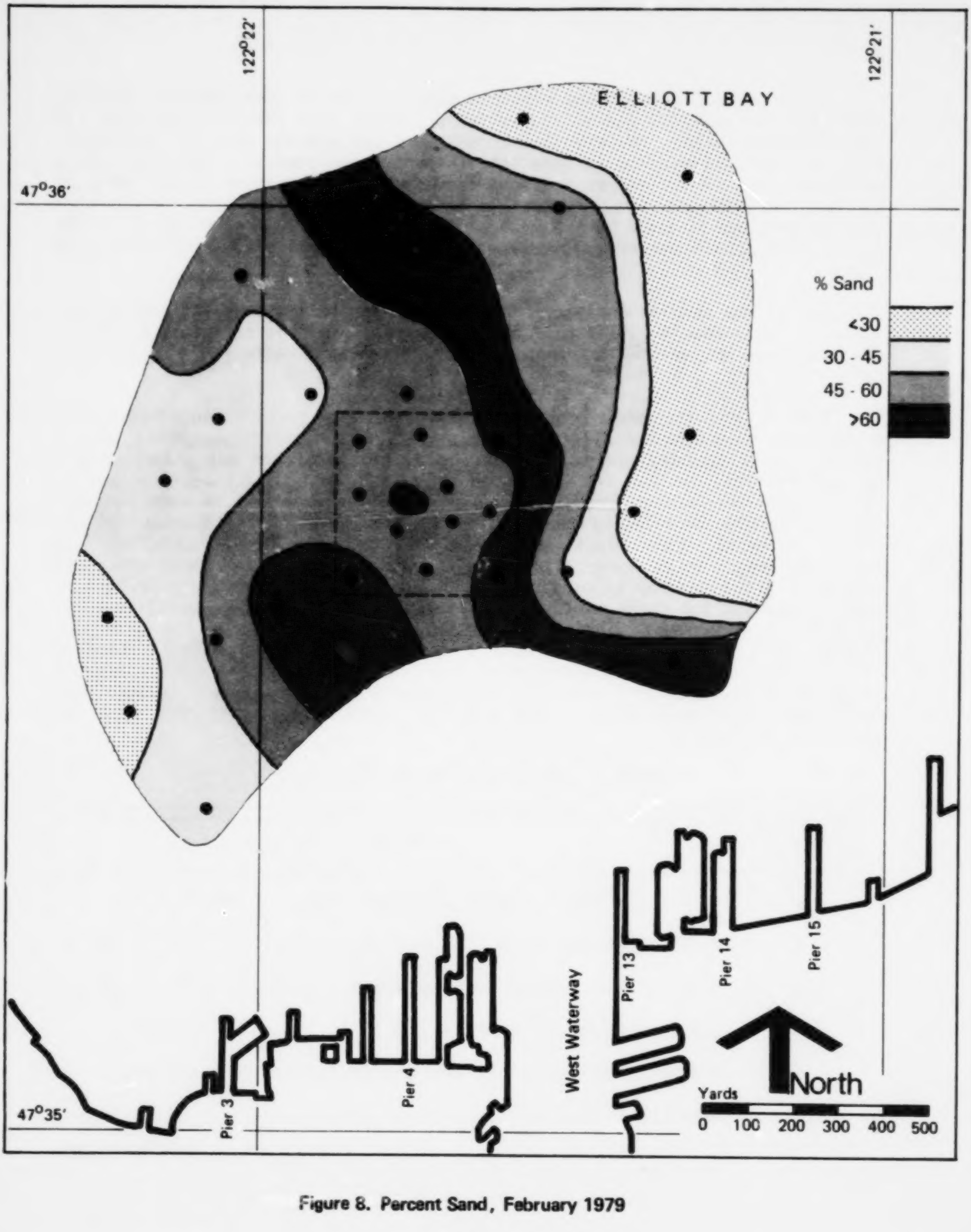


Figure 8. Percent Sand, February 1979

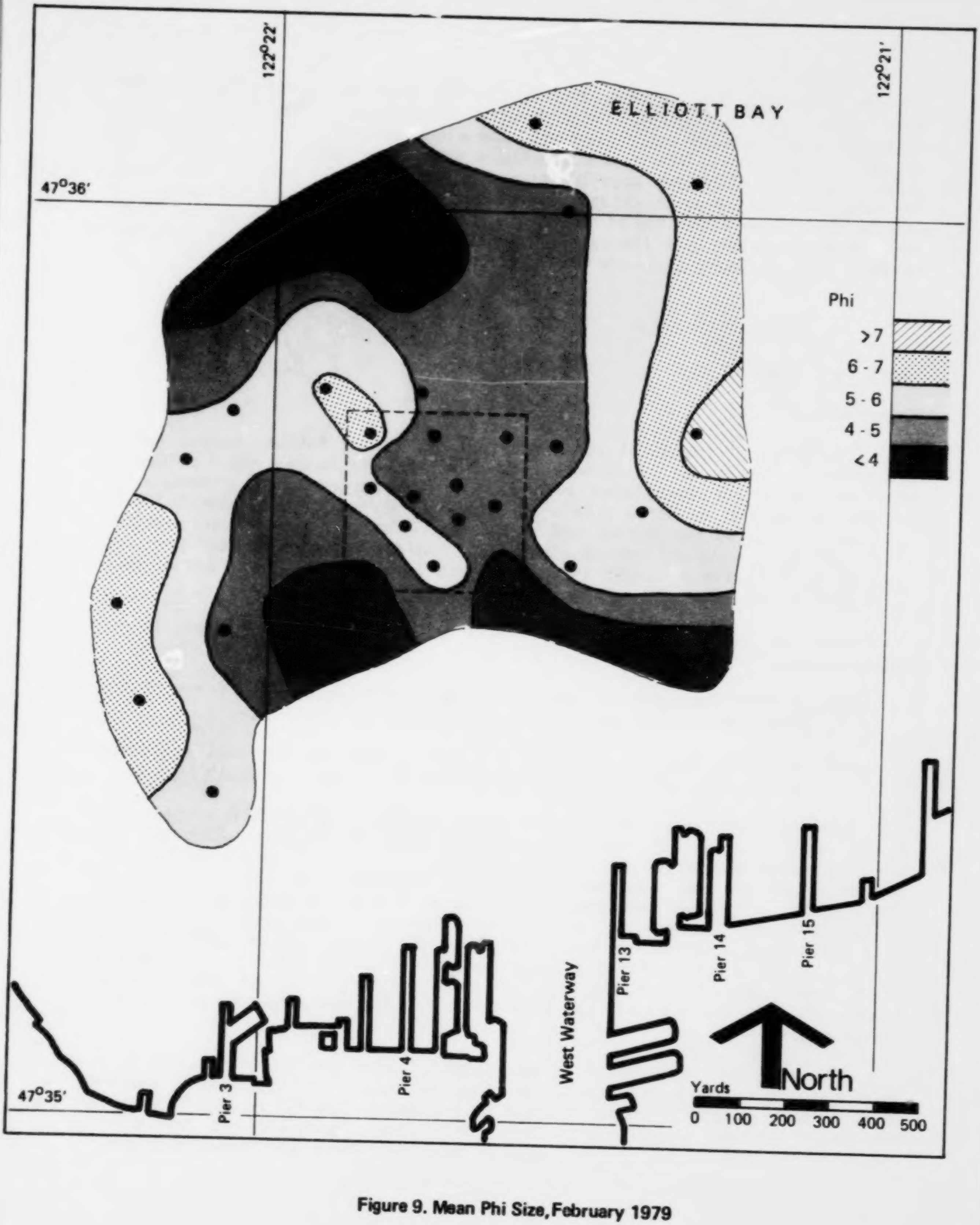


Figure 9. Mean Phi Size, February 1979

By comparing the sediment types with the physical descriptions of the cores, some interesting relationships were established. Types a and b sediment are almost always associated with the black sediment found in and around the grid area, while type f was usually a gray-green sediment from the same area. Type g sediment was usually associated with greenish gray or dark brown sediments and was never associated with black sediments. Type c sediments are mainly associated with samples that were a mixture of greenish gray and black or with other colors not frequently seen. No major association was observed between other sediment types and core descriptions.

PCB RESULTS

Short-Term Study

Water Column

Time plots of the total PCB concentrations during two monitoring events are shown in Figure 10. In general the data indicate rather rapid pulses of high concentrations associated with each of the barge-dumping events. The highest values (up to three orders of magnitude from background levels) were observed at the bottom depths and were associated with particulate matter. After each pulse, the ambient concentrations rapidly returned to near pre-dump conditions, but with a slight increase in PCB levels shown by the end of each daily monitoring period.

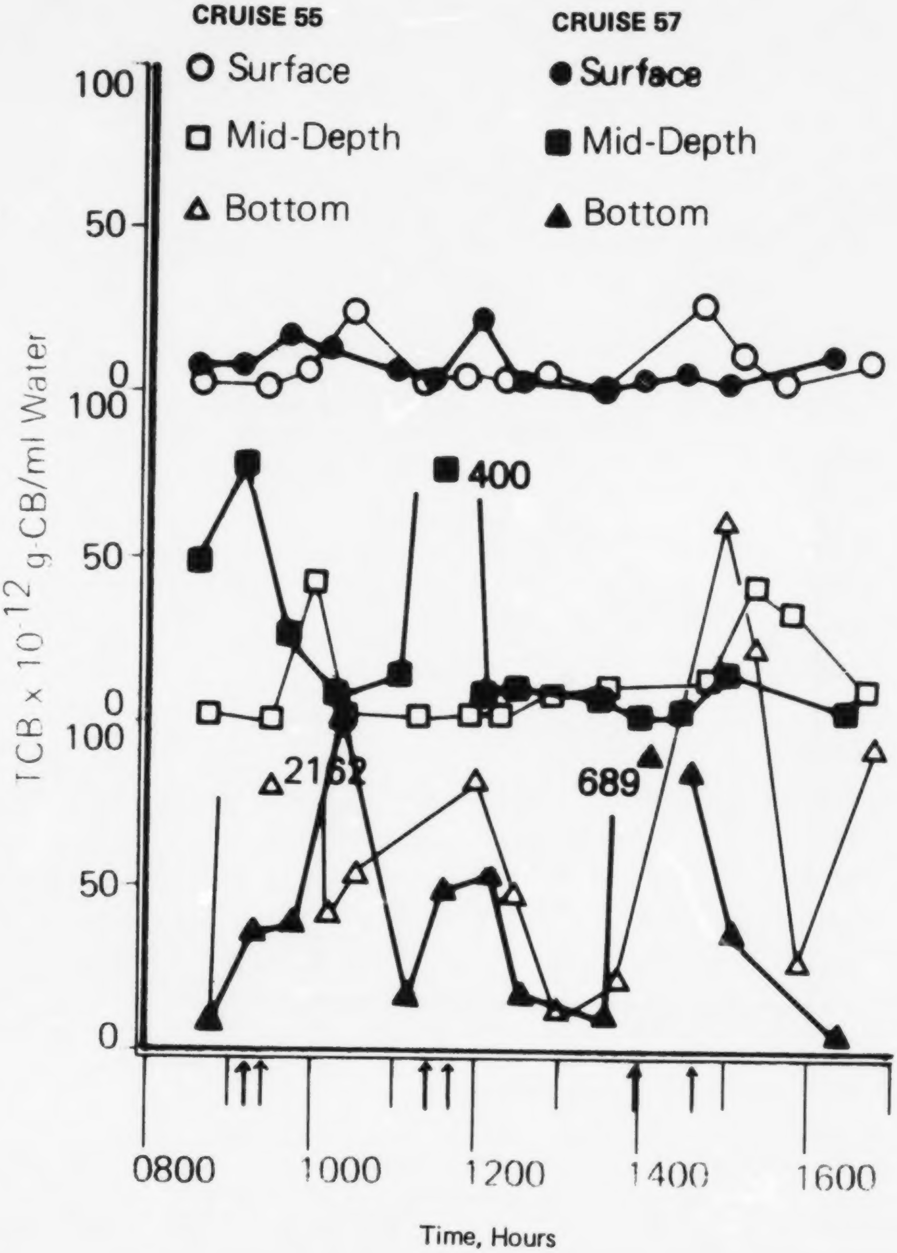
Since the water column within Elliott Bay is normally stratified, it was deemed appropriate to:

(1) examine the vertical profile of PCB concentrations in terms of the hydrographic characteristics of the sampling site;

(2) determine whether residues originated from the disposal operation were maintained and distributed primarily at specific depth layers.

The data indicated that within the depth strata sampled, the highest PCB levels were observed at the surface. This gradient suggests that the low salinity water discharged by the Duwamish River is a major source of contamination within the bay. However, this depth dependence was not statistically significant, which was consistent with the near vertical uniformity observed in the salinity, temperature, and density profiles. These observations agree with the trends observed in earlier studies (Pavlou and Dexter, 1979; Clayton *et al*., 1977). Plots of the average concentrations of PCBs versus time for whole water samples over the post-disposal monitoring period are shown in Figure 11. Similar behavior was also exhibited for the suspended particulate matter samples.

The data from all depths at each station were treated statistically to determine the existence of spatial and temporal patterns. Although it appears that there is a general temporal trend toward decreasing PCB concentrations in the water at each station with time, only levels measured two days after cessation of dumping were significantly different from all subsequent post-



Arrows indicate approximate times of dump episodes.

Figure 10. Plots of Whole Water Total PCB (TCB) Concentrations at Buoy Site (Sta. 6) Versus Local Time

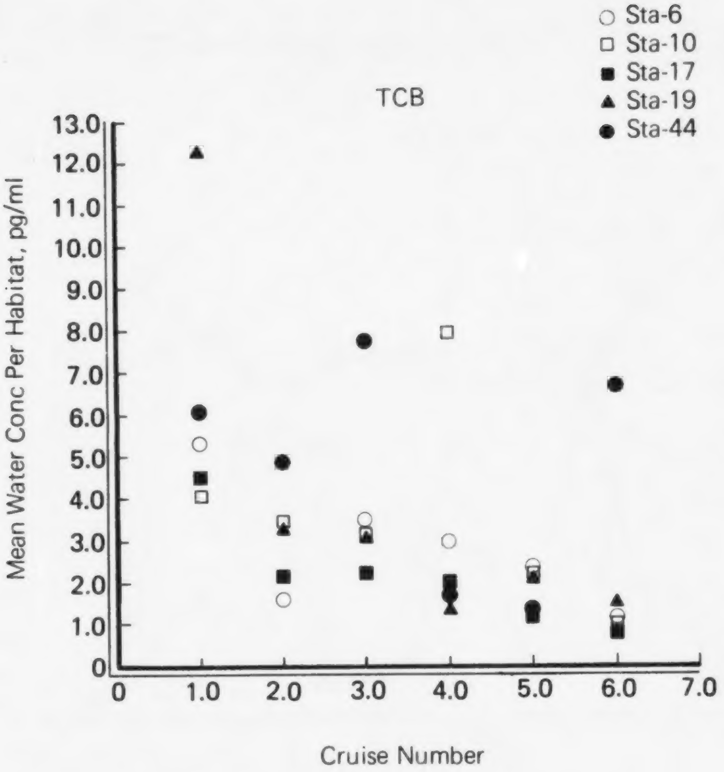


Figure 11. Plots of Mean Habitat TCB Concentrations Versus Time for Whole Water (Post Disposal Cruise Series)

disposal measurements. The mean PCB value for all stations measured two days after dumping ceased was approximately 7 ppt and was higher than had been observed in the past. Comparisons between stations within each cruise did not yield significant differences, although the levels at the east side and Duwamish River (19 and 44) were often higher than the other three stations, probably reflecting the input from the industrialized Duwamish River and Seattle waterfront.

Sediments

The concentrations of PCBs observed in the river sediments, providing the source of material for the disposal project, ranged between 0.01 and 7.0 μg PCB/g dry sediment (ppm). A profile of these values over the stations sampled is presented in Figure 12. Considerable spatial variability was observed with high levels associated with a rather narrow band of highly contaminated sediments towards the northern part of the dredged channel. Both up and downstream, the levels decreased fairly regularly. The mean PCB concentration throughout the section of the river sampled was 2.0 ppm.

PCB concentrations within the sediments of the disposal site and reference areas before dumping showed a pronounced gradient of significantly higher levels in the east (mean about 0.3 ppm) and central portions (mean approximately 0.2 ppm) of the bay and decreasing to the west (mean about 0.03 ppm). Although in general this is consistent with the deposition pattern of contaminated sediments discharged from the Duwamish River as observed in previous studies (Pavlou and Dexter, 1979), very pronounced spatial inhomogeneity was observed, with values in the disposal grid ranging from about 0.1 ppm to as high as 1.7 ppm.

As has been discussed previously (Hafferty et al., 1977), characterizing environmental distributions of PCBs through measurements of the relative mass fractions, F_N, of the individual PCB components provides a useful technique for assessing the dispersal of these chemicals and tracing their source. The characteristic F_N distribution, or "fingerprint," of the river sediments is shown in Figure 13 as a plot of F_N versus the chlorine number, N. The values were generated by determining the relative concentrations of the CB components grouped according to the number of chlorine atoms, the N-CB, averaged over all the river stations. While the pentachlorobiphenyl (5-CB) residues predominated, significant quantities of lower chlorinated biphenyls were observed. In particular, the trichlorobiphenyls (3-CB) averaged about 20 percent of the total.

Although there was little spatial variability in the F_N distributions of the river samples, substantial differences were noted in the F_N profiles for stations within the disposal grid prior to disposal. In spite of the variability in the F_N profiles, all river samples were enriched in lower PCB components, specifically the 3-chlorobiphenyls compared to the background grid sediments. This pattern provided an effective discriminator between the background sediments at the disposal sites and the dredged material from the river.

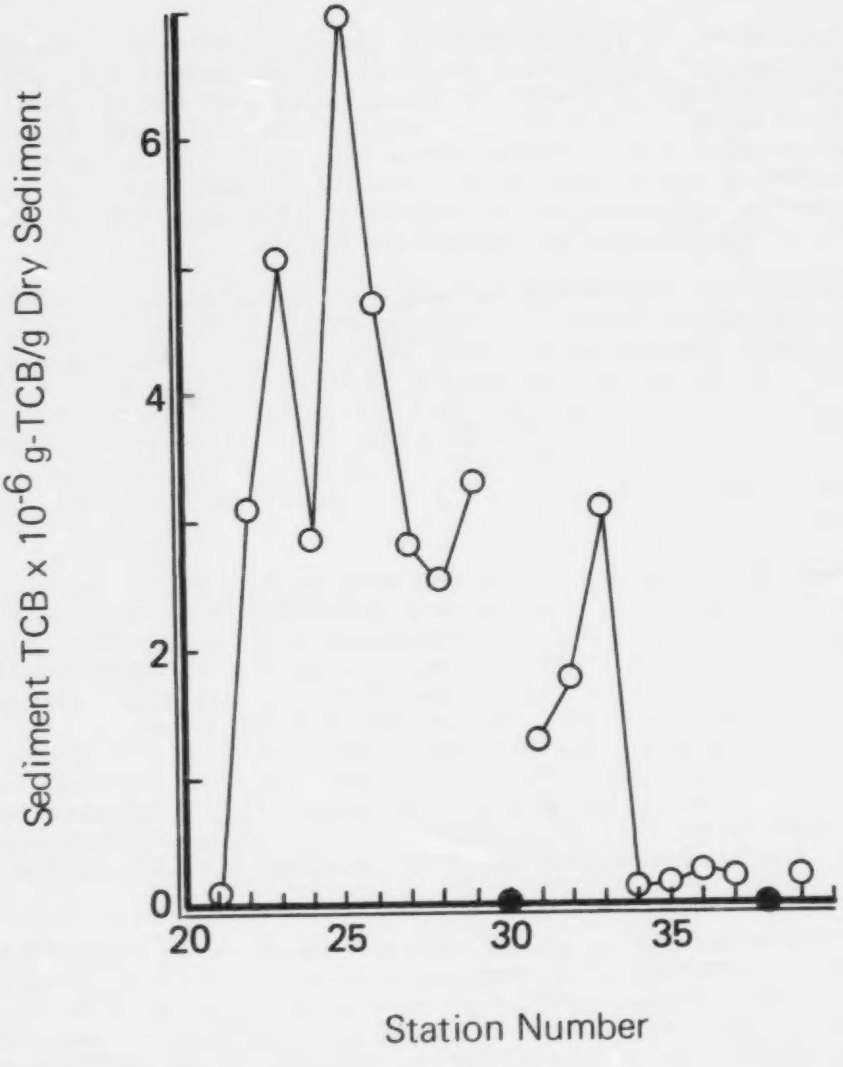


Figure 12. Plots of the Total PCB (TCB) Concentrations in Sediments Versus Relative Distance (Station Number) Within the Duwamish Site

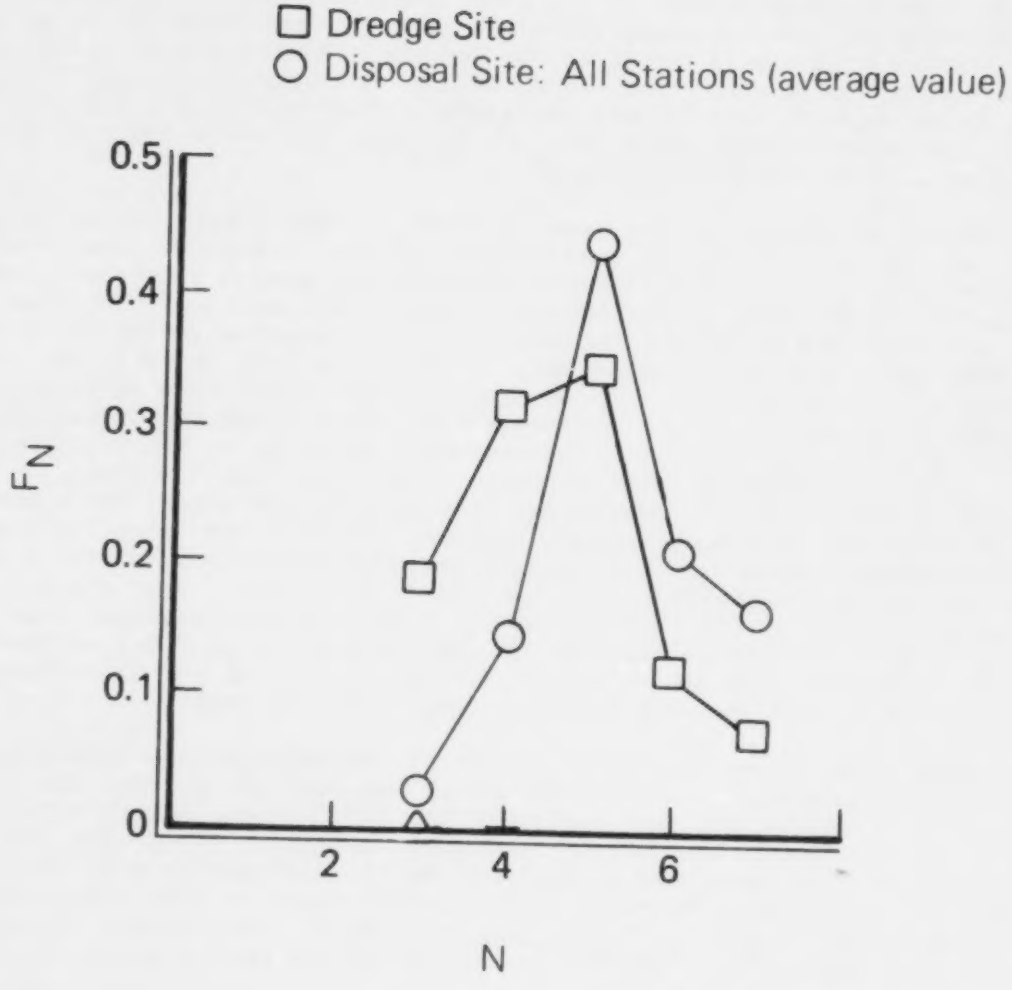


Figure 13. Plots of Relative Mass Fraction, F_N, Versus Chlorine Number, N

To facilitate visualization of the general spatial and temporal trends of PCB residues in the disposal and reference zones during the pre- post-disposal monitoring, three-dimensional histograms of PCB concentrations were constructed (Figures 14 and 15). Inspection of these histograms indicates that after disposal the highest concentrations were encountered at the central section of the station grid with values diminishing roughly radially away from the center. Based on this feature, the grid stations were sorted into three groups consisting of the corner, side, and central stations. The mean values for each group, including the reference stations, plotted as a function of time (sequential cruise number), are shown in Figure 16.

By examining these data, it is clear that there was a significant increase in the PCB concentrations within the upper horizon (upper 10 cm) at all grid station groups as a result of the disposal. No significant change was noted during post-disposal monitoring at the reference stations. Although there is an apparent trend toward increasing concentrations at the grid site during the later cruises (especially at the side and corner groups), these increases were not statistically significant.

PCB concentrations in the lower horizon (sediment depth >10 cm in the core) of the grid during the post-disposal period showed the same general behavior. The statistical analyses of temporal and spatial trends were based on the trichlorobiphenyl concentrations to provide the most sensitive discrimination between the background sediments and those deposited during the disposal operation. Similar to the residue levels in the upper horizon, the lower horizon of the central station group showed a significant increase in CB concentrations immediately after no changes were noted in subsequent samplings. In contrast, the lower horizon of neither the side nor corner groups increased significantly immediately after disposal, indicating that the depth of the original dredged materials deposit was less than 10 centimeters thick around the periphery of the disposal zone. However, the data from later field collections showed a trend toward increasing chlorobiphenyl concentrations within the lower horizon sediments at the side and corner groups. After six months, the lower horizon at the side stations had reached concentrations significantly higher than those observed in the background (surface) sediments. Similar trends were seen at the corner stations, but a significant increase in lower horizon concentrations did not occur until the ninth month.

Comparisons of the mean 3-chlorobiphenyl concentrations for each station group within cruises correlate with these temporal trends. For the first month after disposal, the lower horizon of the central group had significantly higher residue levels than the corresponding horizon at either the side or corner groups. By three months, however, the side and central groups were no longer significantly different. At nine months, none of the three station groups were significantly different from each other. These trends are apparent in the "leveling" of the histograms with succeeding cruises as shown in Figure 15.

In summary, the overall spatial and temporal features of the PCB concentrations suggest that the sediments deposited at the disposal site were not stabilized during the monitoring period. They were slumping from the center of the grid to the periphery. The absence of significant reduction in PCB

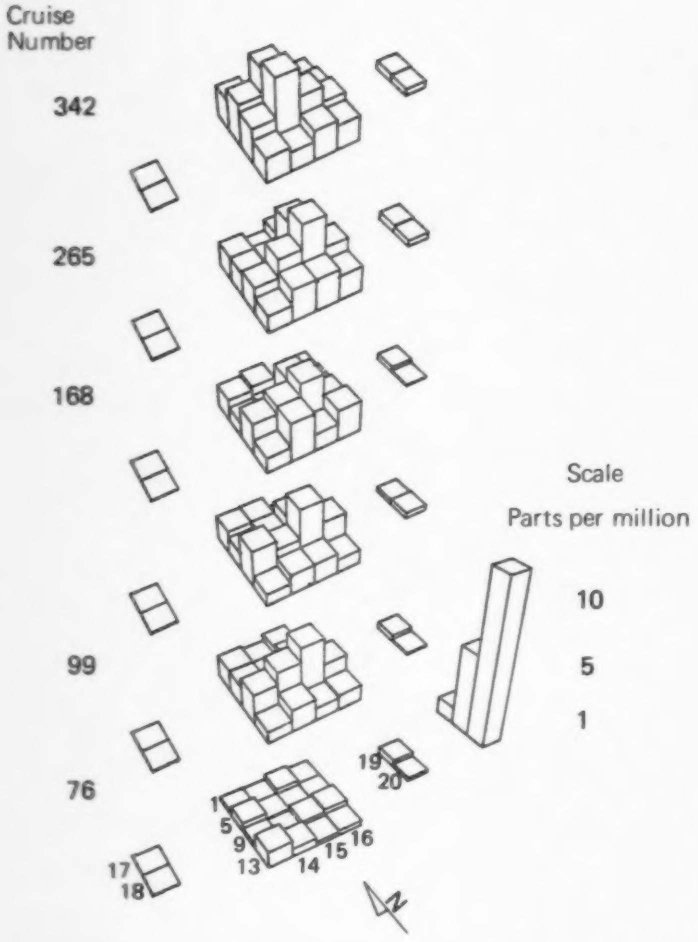


Figure 14. Three Dimensional Histogram of the TCB Concentrations in Sediments for the Upper Horizon

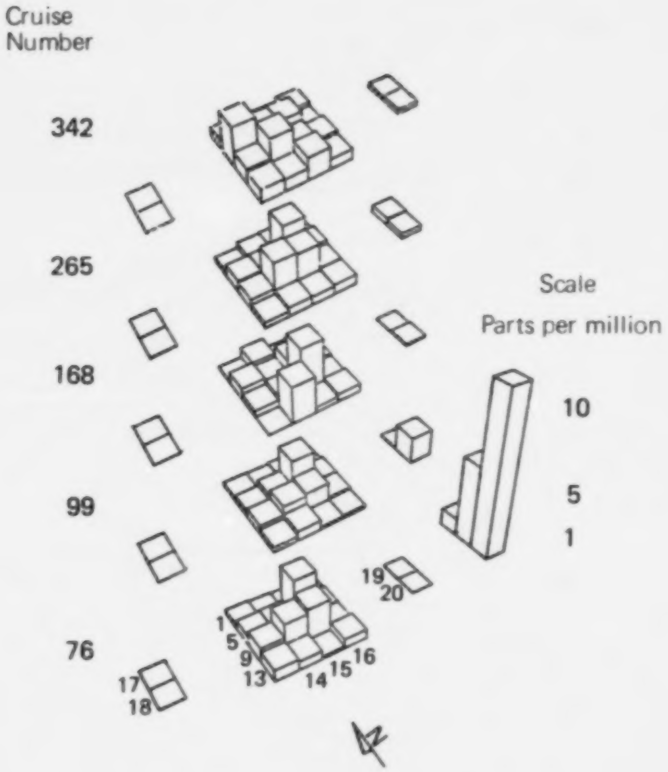


Figure 15. Three Dimensional Histograms of the TCB Concentrations for the Lower Horizon

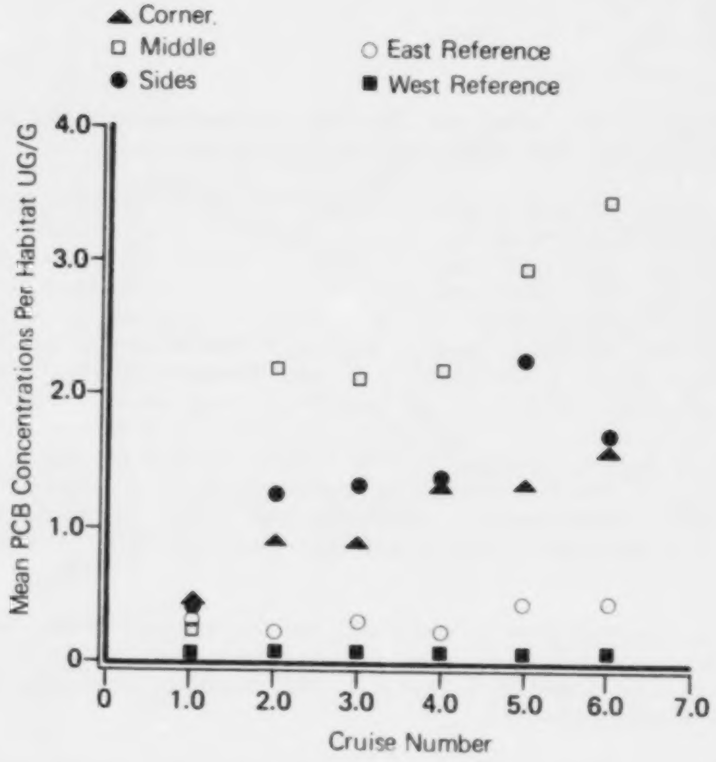


Figure 16. Plot of Mean Habitat PCB Concentrations Versus Time for Sediment in the Upper Horizon

levels in the grid site indicated that no major resuspension or bed-load transport of bottom material had occurred during the monitoring period. This is not unreasonable if one considers the weak and variable velocity field along the bottom of Elliott Bay.

Mass Input Computations

The total amount of PCB deposited in Elliott Bay as a result of the disposal operation was estimated as follows. An average value of 2.0×10^{-6} g PCB/g dry sediment was determined from the PCB data obtained in the river sediments before dredging. The total volume of the sediments dredged was about 1.1×10^5 m^3. Assuming a wet density of 1.3 g/cm^3 and 50 percent water content by weight, a value of 1.4×10^5 g PCB was calculated to have been deposited in the sediments at the disposal site in Elliott Bay.

Long-Term Studies

Sediments

At the present time, data for the PCB concentrations in the sediments from the February and May 1979 samplings are available.

A contour plot of the concentrations of total chlorobiphenyl (t-CB) residues in the surface sediments, combining both sets of data, is shown in Figure 17. Although replicate samples were collected at a few stations, only the highest observed PCB value was used in generating these contours to provide a "worst case" comparison. The general distribution of PCBs agrees with that observed in the previous study. High PCB levels are noted in the vicinity of the disposal site, corresponding to the dredged material deposit, while away from the disposal site, the distribution indicates a similar trend of decreasing PCB concentrations from east to west. Interestingly, the samples do not indicate a major increase in PCB levels associated with the mouth of the West Waterway, a feature noted in previous studies (Pavlou and Dexter, 1979). This latter observation, however, may result simply from the high variability of the sediment levels and the low sampling intensity near the river mouth.

Similarly, a contour plot (Figure 18) of the highest t-CB levels observed in any core from each sampling station, irrespective of depth in the core, shows essentially the same distribution, but extends eastward--the apparent influence of the dredge material.

Comparisons of these t-CB values alone, however, were insufficient to precisely delineate the dredged material deposit, particularly toward the east where high levels were observed in many of the samples.

As discussed earlier, the PCBs in the dredged material were noted to be relatively enriched in the lower chlorinated CBs. Therefore, the concentrations of the trichlorobiphenyl (3-CB) residues in the sediments were compared. As expected, the 3-CB distribution (Figures 19 and 20, for surface and highest values, respectively) shows a spatial extent of the dredged material in close agreement with that anticipated from the results of the previous study and the recent bathymetric survey.

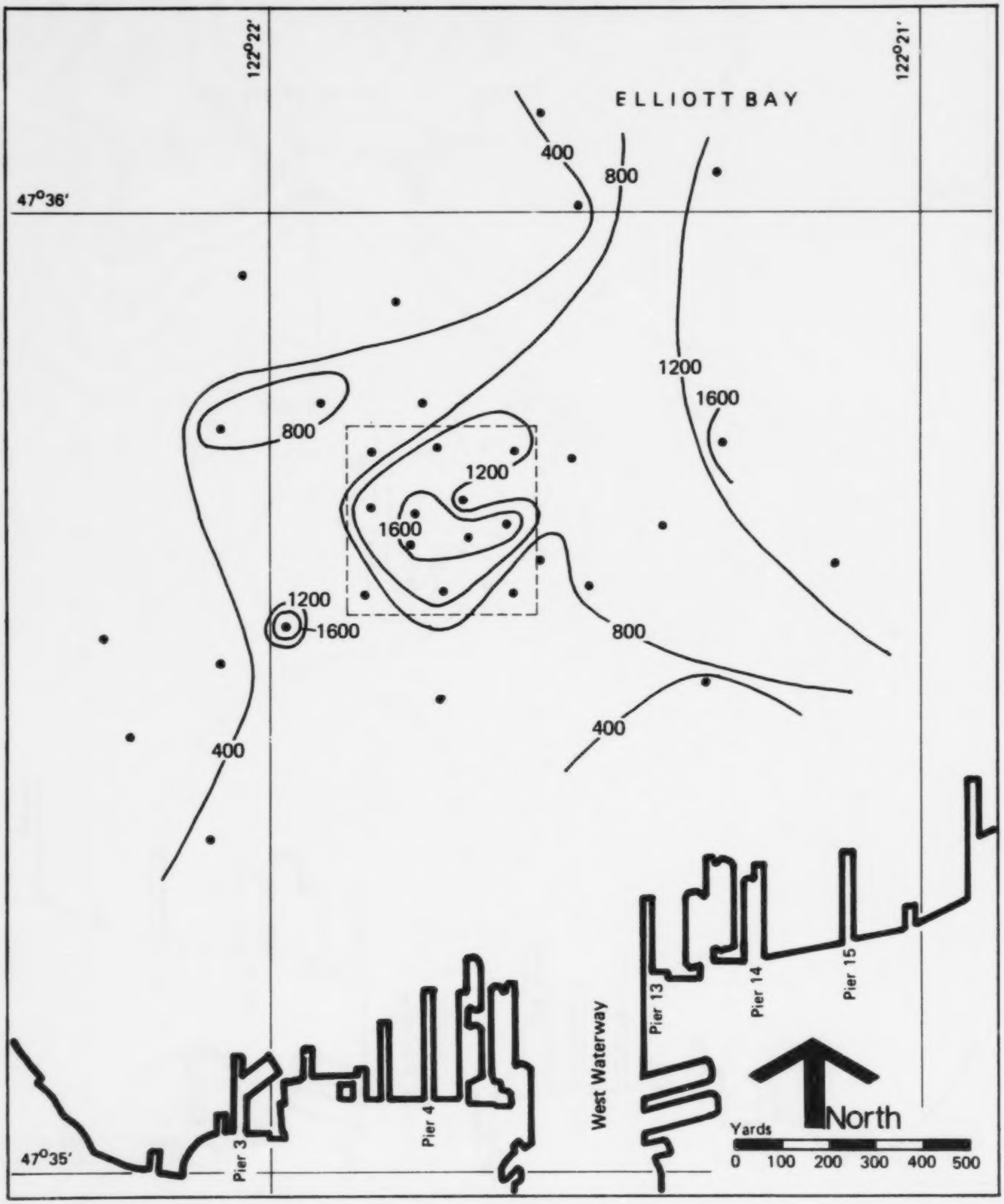


Figure 17. Contour Plot of the Concentrations of TCB Observed in the Surface Sediment Horizon (In Units of ng t-CB/g dry sediment)

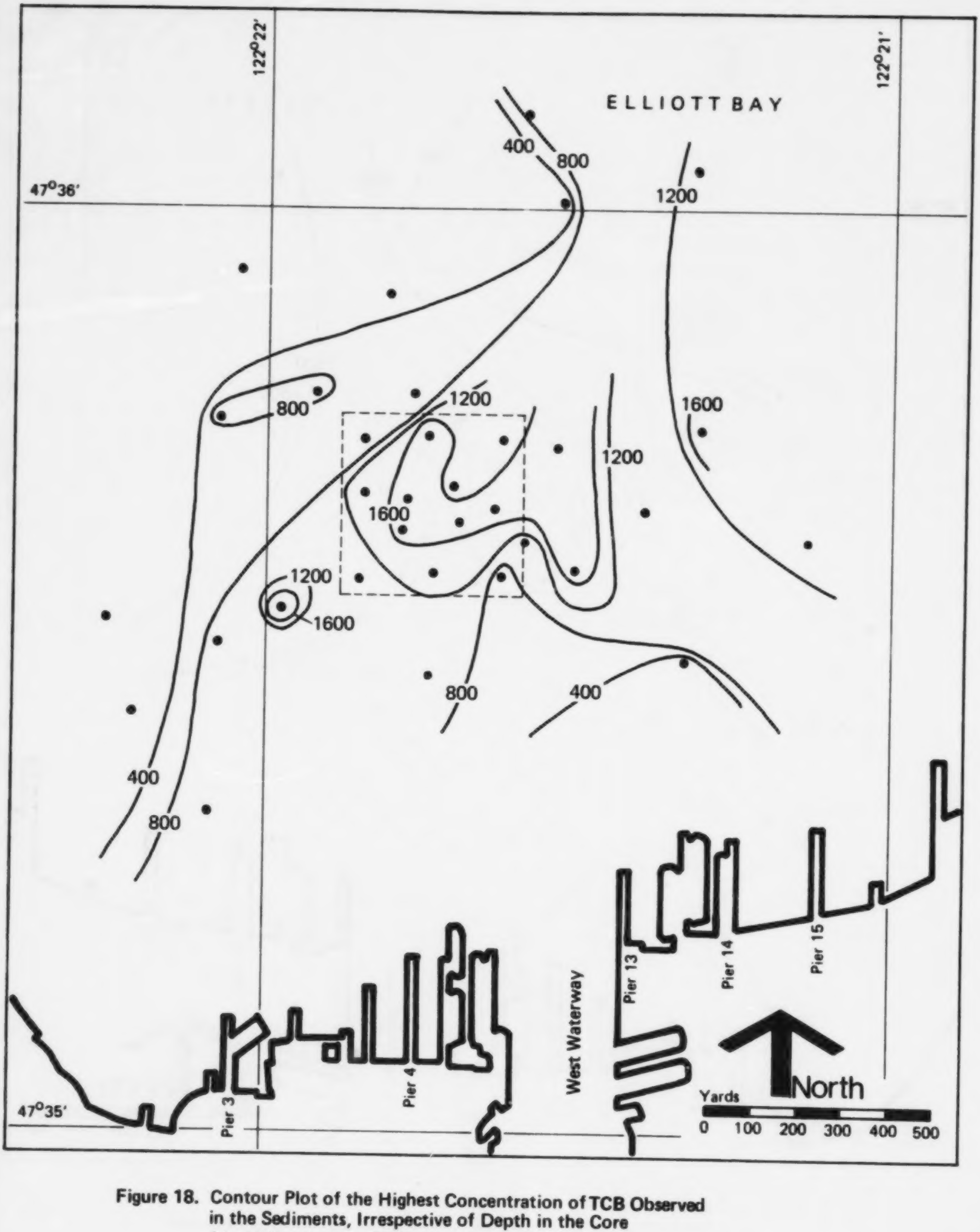


Figure 18. Contour Plot of the Highest Concentration of TCB Observed in the Sediments, Irrespective of Depth in the Core (In Units of ng t - CB/g dry sediment)

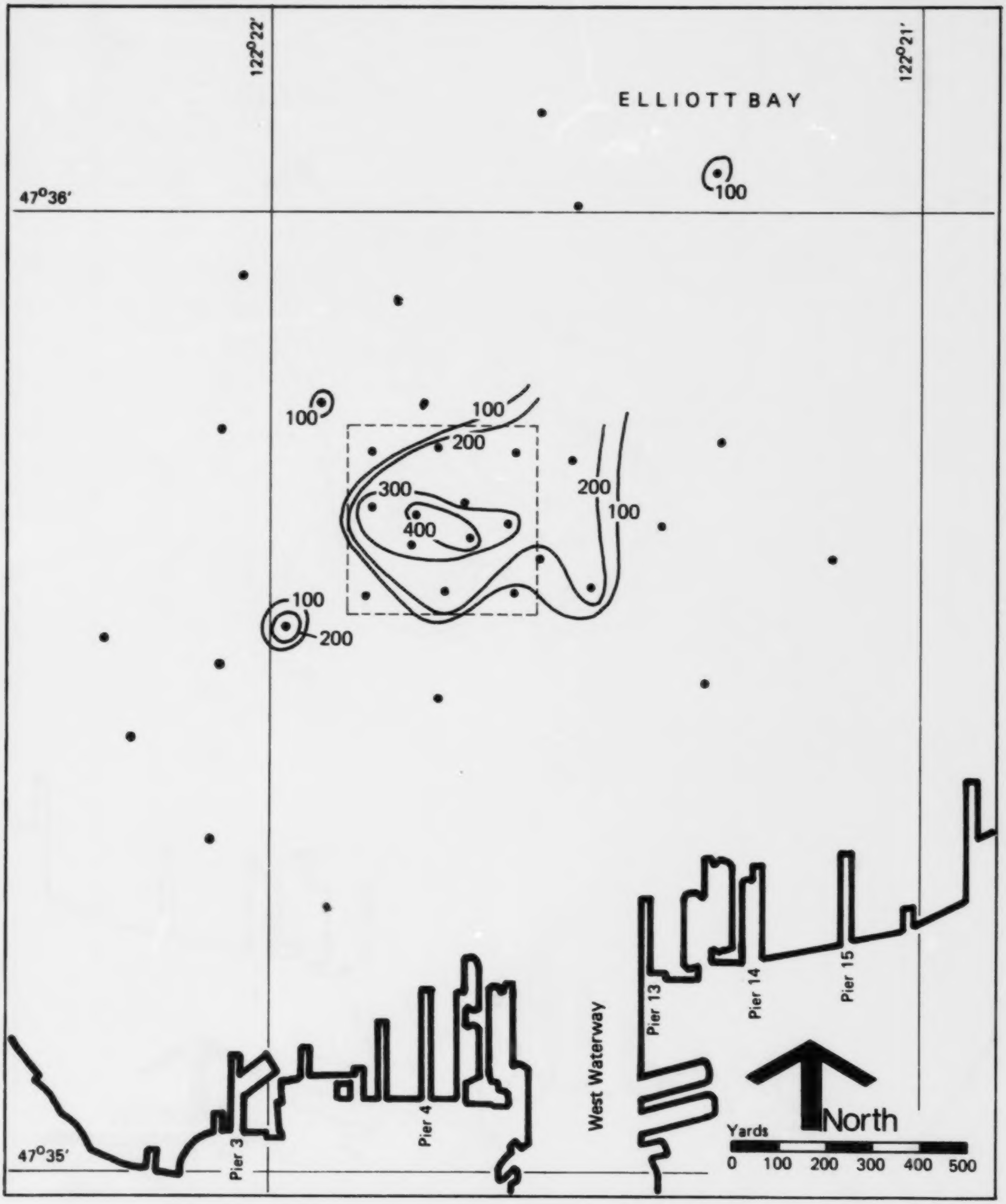


Figure 19. Contour Plot of the Concentrations of 3 - CB in the Surface Sediment Horizon
(In Units of ng 3-CB/g dry sediment)

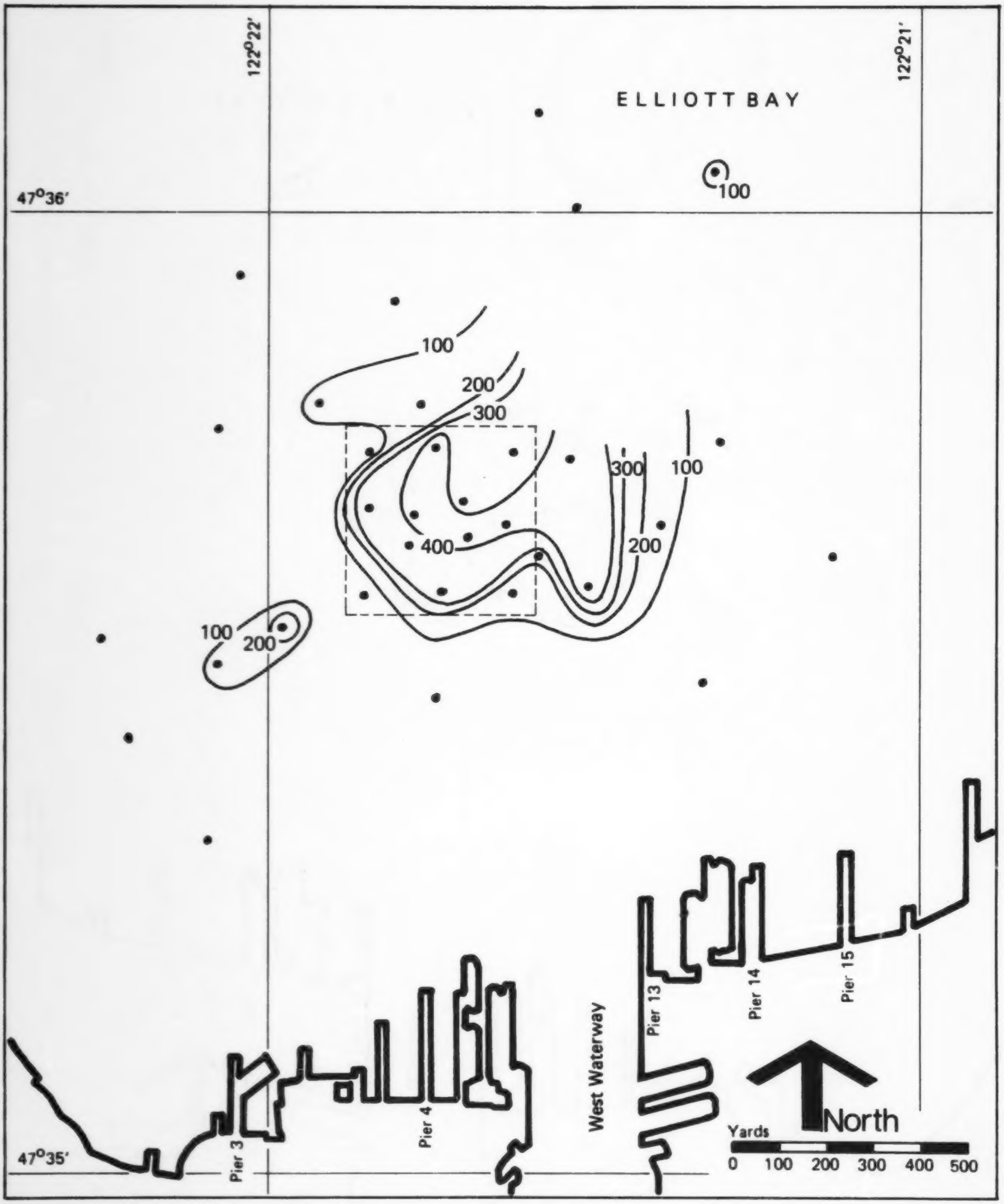


Figure 20. Contour Plot of the Highest Concentration of 3-CB Observed in the Sediments, Irrespective of Depth in the Core (In Units of ng 3-CB/g dry sediment)

Tracing the Distribution of Dredged Material

As noted above, a number of values from stations outside of the original disposal monitoring grid also had both high 3-CB and high t-CB residue concentrations. In addition a number of samples from both inside and outside the grid area had intermediate 3- and t-CB levels. Both factors preclude a clear distinction between dredged material deposits and native sediments based on differences in their PCB content only. It was noted, however, that the distributions of certain sediment types, and the high -3-CB levels closely corresponded. Therefore, a histogram was constructed (Figure 21) to relate the levels of 3-CB in each sample to the corresponding sediment type. On the basis of this histogram, two dominant groups of sediments can be delineated; one group characterized as sediment types c and g with low 3-CB concentrations, the other by f, b and f/b sediment types and higher 3-CB levels. The latter group is considered to be representative of at least the majority of the dredged material deposit. The remaining sediment samples fell in intermediate ranges with no clear discriminator available at this time.

Those stations having sediments that clearly met the criteria for dredged material are indicated in Figure 22, which also shows the approximate depths of the dredge-material deposit at each location as determined by the length of the core from the surface to the bottom of the lowest horizon consisting of dredge material.

While differences in the sampling plan preclude direct point-to-point comparison of the PCB levels at each location, a comparison of the overall levels is possible. The mean t-CB concentration from the average of all samples of dredge-material observed in the original study was about 1.7 ppm, while the corresponding mean 3-CB concentration was 0.29 ppm. A similar averaging of all of the available 1979 data for dredged-material samples yielded 1.9 ppm and 0.4 ppm for the t-CB and 3-CB concentrations, respectively. Considering the variability observed in both the old and newer data sets, these means are undoubtedly statistically indistinguishable.

The distribution of the dredged-material deposit delineated by these results is in general agreement with the supposition that little movement of the deposit or change in the associated PCB levels has occurred since the original disposal operation. There are indications, both in the position of the observed dredged-material sediments and in the bathymetry (see above), that some shifting of the disposal mound to the east or northeast may have occurred over the last few years. The highest PCB values were observed at station 114, just east of the grid, while the west and south grid corner stations (109, 111, and 112) had no dredged-material.

Interestingly, the distribution generally agrees with what would be expected from fluid flow of the dredged material along bathymetric contours, which would have occurred with the slumping and settling noted above. However, it must be recognized that detailed delineation of the spatial extent of the deposit is not possible considering the level of sampling intensity employed and the limitations in the precision of the bathymetric surveys. In addition, since no sampling was performed outside the grid during the initial study, it is difficult to determine when the eastern deposits originated.

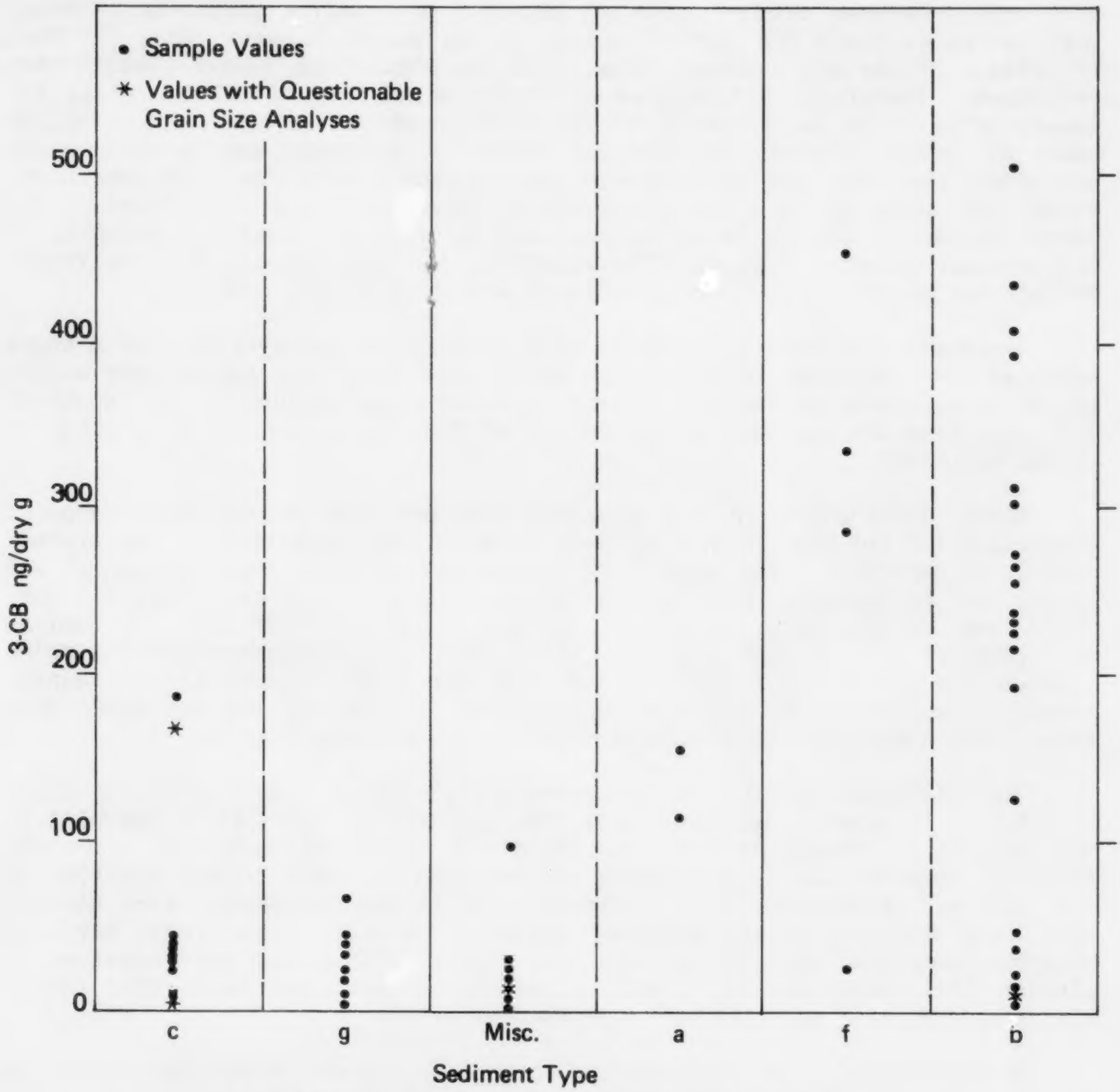


Figure 21. Histogram Relating the Concentrations of 3-CB in the Sediments to the Sediment Type

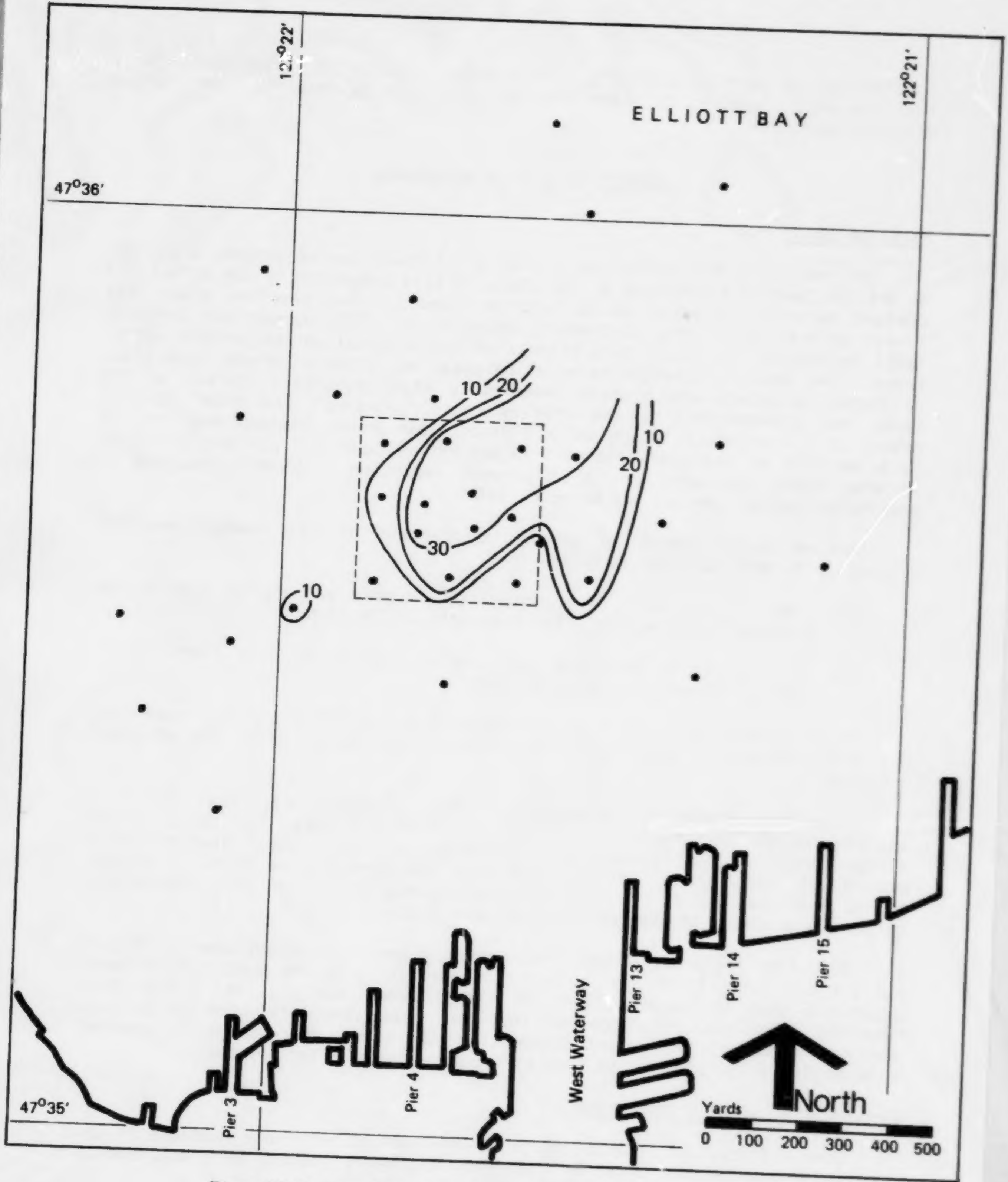


Figure 22. Contour Plot of the Approximate Depth of the Dredge - Material Deposit (In cm)

Considering the general stability of the major deposit, it seems most likely that the major features of these distributions were generated by the initial disposal operation.

IMPACTS TO BENTHIC MACROFAUNA

Baseline Study

The rationale for conducting a biological study was to determine whether or not the benthic macrofauna at the disposal site responded to the effects of dredged material disposal on a long-term basis. The baseline biological investigations (U.S. Army Engineers, Appendix G, 1978; Harman and Serwold, 1978) documented the short-term effects of the disposal on the benthic macrofauna. The impact of dredged material disposal was evident through reductions in faunal abundance and biomass immediately after disposal. Burial of the fauna was hypothesized as the primary cause. Although the total spatial extent of the disposal impact was not discernible due to limited sampling, it is known that at least an area of 0.13 km^2 experienced a 21 percent reduction in mean faunal abundance and a 25 percent reduction in biomass compared to predisposal values (Harman and Serwold, 1978).

Recolonization phases of benthic macrofauna at the dredged material disposal site were reported as follows:

(1) summer recruitment of opportunistic benthic macrofaunal species and annuals (three months after disposal), followed by

(2) an increase in predatory polychaetes and a decline in opportunistic species (Harmon and Serwold, 1978).

These authors stressed that the composition of the biological community had not returned to predisposal conditions nine months after the disposal occurred.

U.S. Army Engineers, Appendix G, 1978, provided additional numerical analyses to the data reported by Harman and Serwold (1978). The impact of the dredged material disposal was greatest at the central disposal stations, with mean faunal density and biomass remaining low through nine months. In contrast, many stations at the margins showed greater values for mean abundances, biomass, and number of species than the reference stations.

U.S. Army Engineers (1978) summarized the overall dredged material disposal project for Elliott Bay. Discussion pertaining to the benthic macrofauna suggested that the major effect of the disposal was physical rather than chemical. The authors suggested that the most obviously impacted central stations suffered no permanent damage since the mean number of species present climbed from a low of three to twenty-five nine months after disposal.

Present Biological Characteristics

Some difficulties with appropriate selection of reference stations in the short-term studies, as cited by U.S. Army Engineers (1978), were overcome by using a different sampling design for the long-term studies, as described earlier in this paper.

Biological data analyses in the present study were different in many respects to earlier works. Harman and Serwold (1978) analyzed the data primarily by calculation of mean values and production of geometric contour plots of the disposal site and reference station to show spatial and temporal data trends.

U.S. Army Engineers, Appendix G (1978) used numerical analyses, emphasizing statistical methodologies, including ANOVA and the nonparametric Kruskall-Wallis test. In this study, statistical treatment of the data includes mapping, cluster analysis, Wilcoxon's two-sample test, spatial autocorrelation, Kendall's coefficient of rank correlation, and multiple regression analysis.

Preliminary analyses of faunal abundances have been conducted on data from both the February and May 1979 cruises. Discussion is limited here to analyses of abundance from the May 1979 cruise and is based on mapping, cluster analysis, and Wilcoxon's two-sample test.

Mapping

Mapping of the biological data was conducted to provide a preliminary evaluation of the spatial trends in taxa abundances. Two approaches to mapping the individual taxa abundances were used:

(1) calculating mean values for each station's three replicates and manually producing geometric contour maps;

(2) dividing the range of abundances for each taxa into discrete subsets and plotting individual replicates.

For purposes of visual representation of biological data, the authors feel that contour mapping may oversimplify spatial patterns due to unknown variability in abundance values and may present a biased picture of abundances. Therefore, the second method of mapping was selected for display since it does not rely on mean values, and does not necessitate interpolation of abundance values between samples. Figure 23 shows the spatial abundance of the polychaete family Capitellidae. The plot suggests that higher abundances occur within and in the immediate proximity of the original sampling grid. Similar tendencies were also found for Axinopsida sericata, the most abundant bivalve; several of the Maldanidae, including Praxiella affinis, P. gracilis, Maldane glebifex, and Euclymeninae; and Paronella spinifera and Eteone sp., polychaetes from the families Paraonidae and Phyllodocidae, respectively. These hypotheses are tested explicitly (below).

Cluster Analysis

Cluster or numerical classification analysis was conducted to combine station replicates (samples) into groups based on similarities in taxa abundances. The purpose of grouping the samples was to form testable hypotheses concerning spatial patterns in taxa abundances relative to the disposal site.

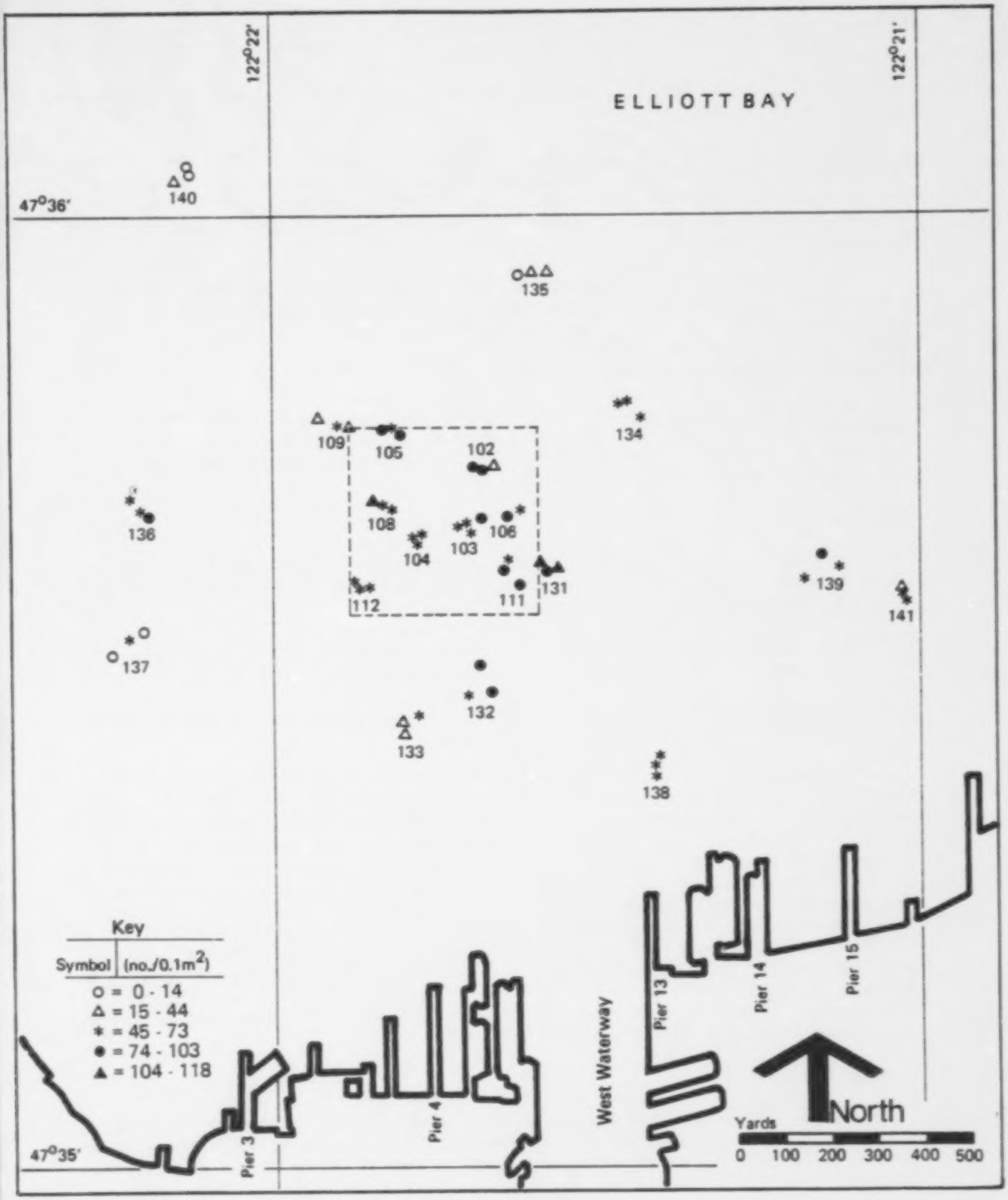


Figure 23. Spatial Distribution of Polychaete Family Capitellidae

Cluster analysis was performed using the CLUSTAN (IC) computer program developed by Wishart (1975). All abundance values were species-total standardized (e.g., Boesch, 1977) prior to calculating similarity indices. The Bray-Curtis Index was used to calculate the "distance" between samples. The Lance-Williams flexible beta combinatorial method was used to calculate distances between groups of samples (Boesch, 1977). These computations were initially run using the seventeen most abundant taxa sampled in the May cruise and considering each station replicate as a separate entity (60 x 17 matrix). These taxa are listed in Table 3. Cluster analysis was also run using these same taxa and their mean abundances from the three replicates at each station (20 x 17 matrix). The results are shown in the dendrogram plots in Figures 24 and 25 for the individual sample abundances and mean sample abundances, respectively. Three distinct station groupings within each of these plots are evident. Figure 26 shows the spatial relationship of these groups of stations/samples relative to each other and the original sampling grid.

TABLE 3. RANKING OF TAXA FROM MAY CRUISE BY ABUNDANCE

Rank	Abundance Taxa	Organism Type[3]	Mean Density[1] (Abundance 0.1 m^2)	Percent[2] Relative Abundance
1	*Axinopsida sericata*	C	270	100
2	Capitellidae spp.	MW	58	21
3	Euclymeninae spp.	MW	51	19
4	*Paraonella spinifera*	MW	36	13
5	*Macoma carlottensis*	C	25	9
6	*Aricidae* cf. *lopezi*	MW	17	6
7	Cossuridae sp.	MW	16	6
8	*Lumbrineris luti*	MW	10	4
9	Amphipoda spp.	CR	9	3
10	*Nuculana minuta*	C	9	3
11	*Prionospio cirrifera*	MW	8	3
12	*Chaetozone setosa*	MW	7	3
13	*Glycera capitata*	MW	4	1
14	*Nephtys ferruginea*	MW	4	1
15	Ostracoda spp.	CR	3	1

[1] Total number of particular taxa abundance for all replicates divided by 60.
[2] Percent relative abundance to *Axinopsida sericata*.
[3] MW-marine worm; C-clam; CR-crustacean.

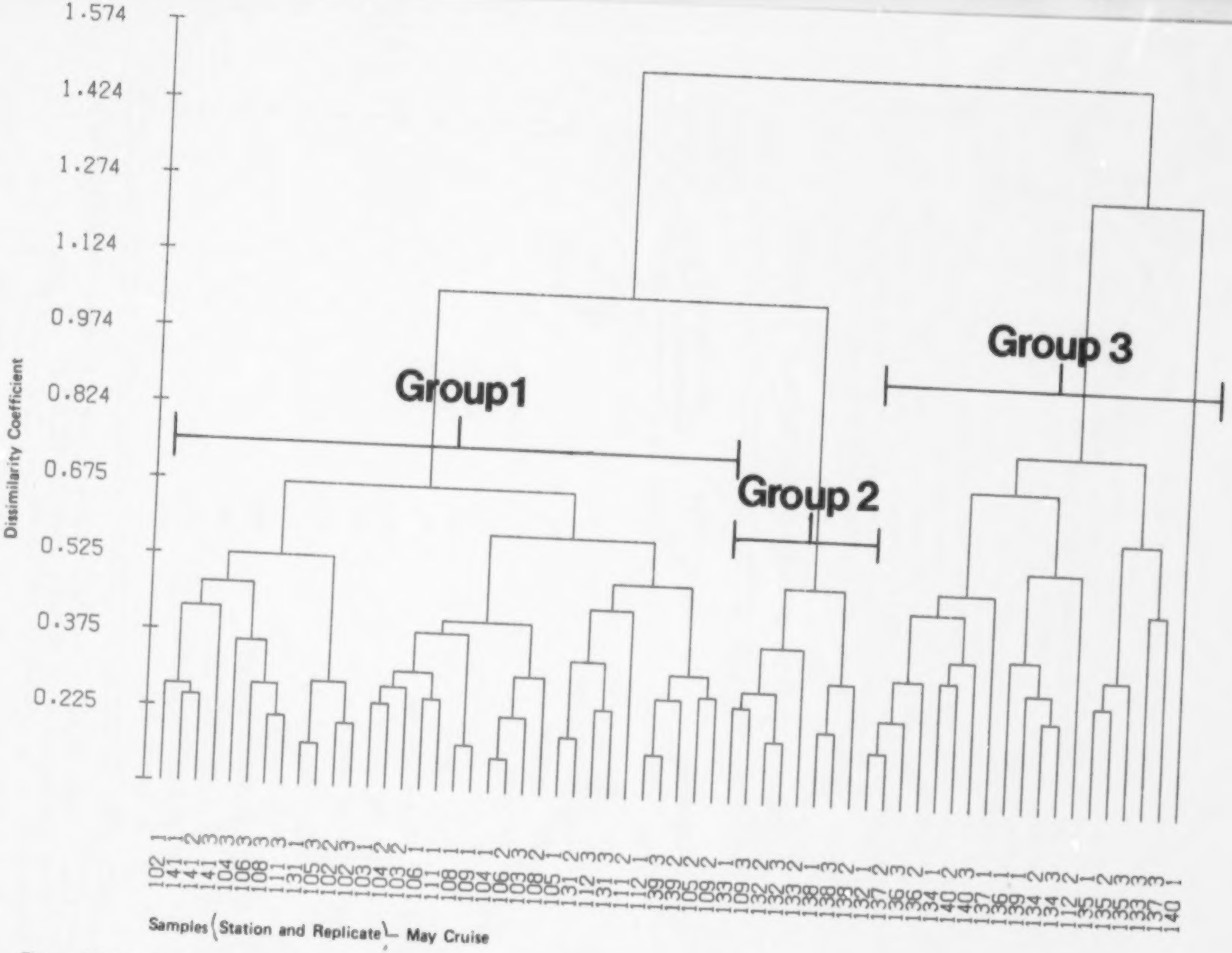


Figure 24. Cluster Analysis of Biological Organism Abundances for May Cruise/Matrix Equals 60 Replicates by 17 taxa

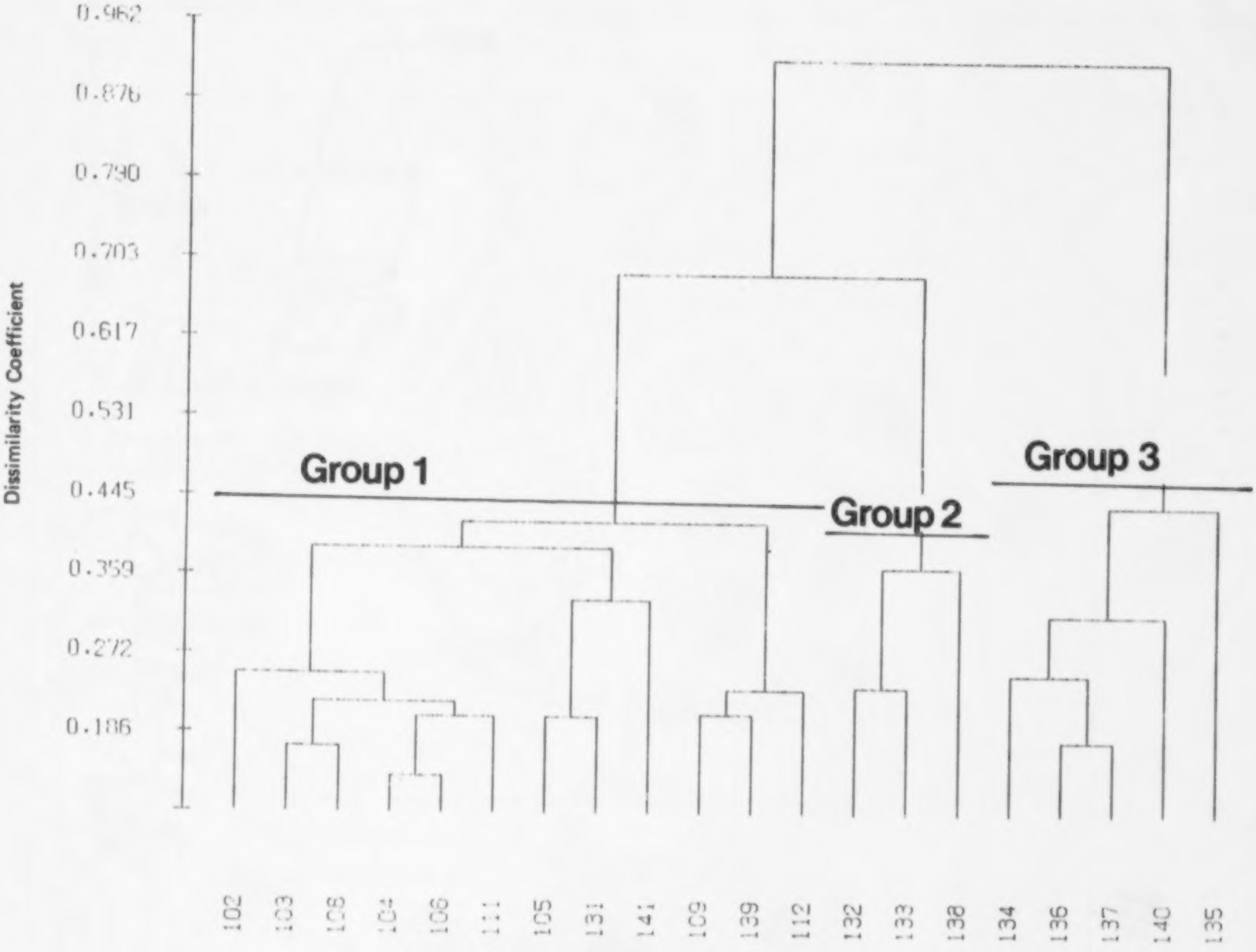


Figure 25. Cluster Analysis of Biological Organisms Mean Abundances for May Cruise/ Matrix Equals 20 Samples by 17 Taxa

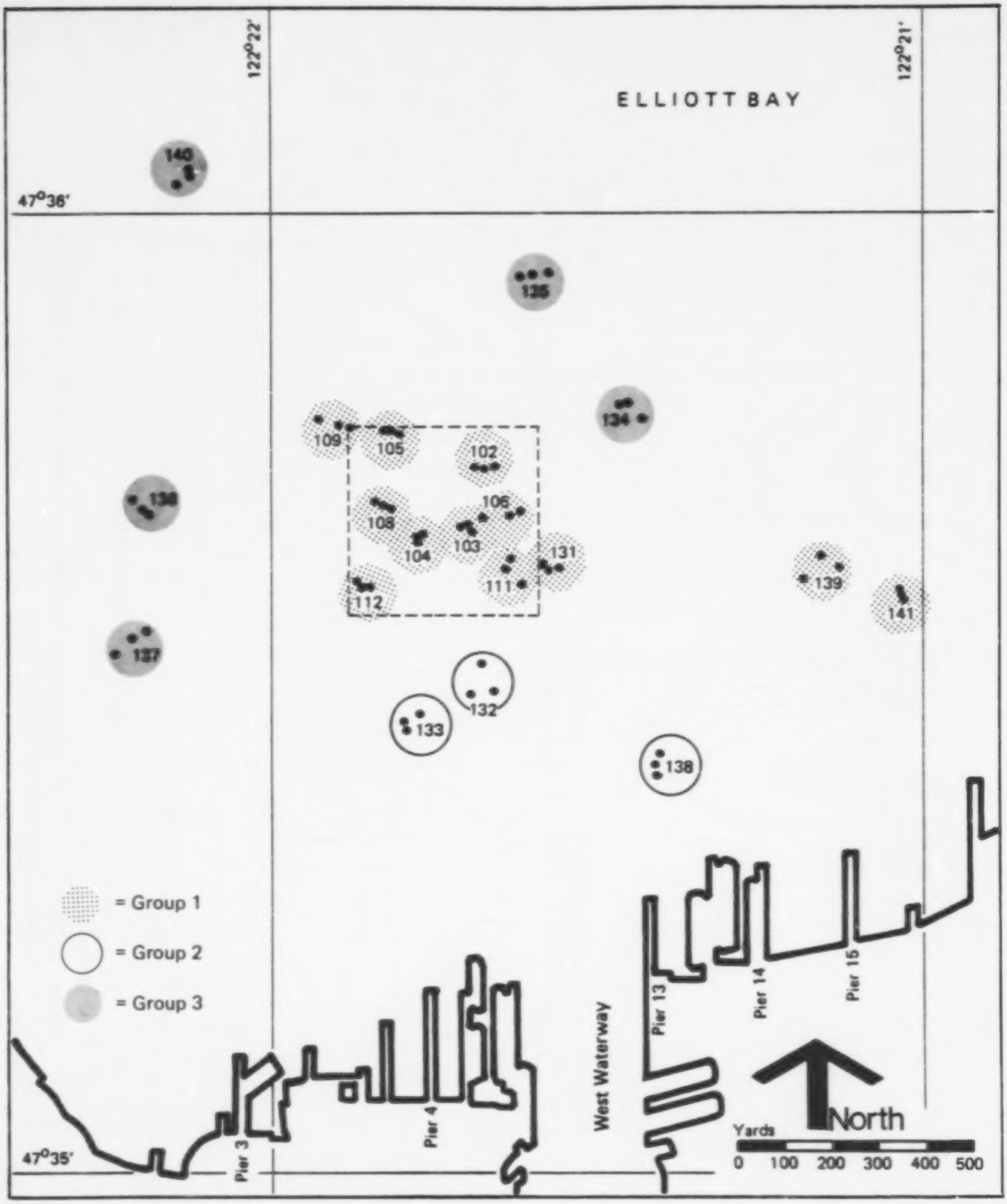


Figure 26. Samples(Station and Replicate)Locations for Biological Sampling May Cruise (as grouped by cluster analysis)

The cluster analysis results suggest that taxa abundances for samples in and immediately around the original sampling grid (group 1) appear to be more to each other than to samples more distant from the grid (groups 2 and 3). This analysis suggests that the disposal site contains a unique assemblage of macrofauna. This hypothesis is tested explicitly (below). Two obvious exceptions to this generalization are stations 139 and 141, which were clustered in group 1.

Wilcoxon Two-Sample Test

The nonparametric Wilcoxon two-sample test was conducted to determine whether or not particular taxa, suggested from mapping and cluster analysis, had statistically significant differences in abundances within and around versus more distant from the disposal area. A nonparametric test was used because it requires no assumptions of normally distributed populations. The Wilcoxon two-sample test was run using a subroutine of the Statistical Analysis System Institute, Inc., 1979, data base management system.

The test was run for seventeen taxa using two different station groupings. The first run used a station grouping which combined groups 2 and 3 from the cluster analysis and compared it to group 1. The second run used an identical grouping except stations 139 and 141 were excluded from group 1 and put into the combined groups 2 and 3 (see Figure 26). This approach compared all stations within and immediately around the disposal site to all distant stations.

The results of the analyses for station groupings are summarized in Table 4. Results were nearly identical for both test groupings. The abundances of nine of the seventeen taxa were different between groups (statistically significant at the 1 percent level). All of these taxa had greater abundances within the grid and in its immediate proximity.

Relationships of Biological Results to Physical/Chemical Results

As previously indicated, the scope of the discussion was to describe results of the mapping, cluster analysis and Wilcoxon two-sample tests. Spatial autocorrelation is presently being conducted with preliminary results supporting the trends shown by the Wilcoxon two-sample tests. The analyses planned for the immediate future will attempt to relate the biological results to the physical/chemical results. These analyses include multiple regression analysis and Kendall's coefficient of rank correlation. Those independent variables considered relevant to the multiple regression analysis include depth, distance from the disposal grid center, PCB concentrations in the sediment and interstitial water, mean sediment size, and percent organic carbon in the sediment.

The physical and chemical data analyses have shown that the sediments at stations in and near the disposal grid consist of dredge material containing unsorted sediments and elevated PCB concentrations. The biological analyses have also demonstrated that taxa abundances are different in the same area as compared to more distant stations. This similarity in results may indicate a long-term physical and/or chemical impact on the biological community. Possible relationships will be investigated as a part of this research project.

TABLE 4. RESULTS OF WILCOXON TWO-SAMPLE TESTS FOR SELECT TAXA FROM GROUPS OF STATIONS[1]

		Group 1 Compared to Groups 2 and 3 Combined[2]		
Difference Does Exist between Groups[3]	Probability	Taxa Abundance Lower or Higher on Original Sampling Grid	No Difference Exists between Groups	Probability
Axinopsida sericata	0.0001	Higher	Nuculana minuta	0.2143
Capitellidae spp.	0.0001	Higher	Prinospio cirrifera	0.2488
Euclymeninae spp.	0.0001	Higher	Glycera capitata	0.0444
Paraonella spinifera	0.0005	Higher	Ostracoda spp.	0.9234
Macoma carlottensis	0.0001	Higher	Laonice cirrata	0.0919
Cossuridae sp.	0.0000	Higher	Nucula tenuis	0.1373
Chaetozone setosa	0.0001	Higher	Aricidae cf. lopezi[4]	0.0237
Nephtys ferruginea	0.0073	Higher	Lumbrineris luti	0.0101
Amphicteis scaphobranchiata	0.0008	Higher		

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[1] Taxa are the same as those used for cluster analysis of May cruise biological data.

[2] See Figure 26 for better understanding. Group 1 includes Stations 102, 103, 104, 105, 106, 108, 109, 111, 112, and 131. Group 2 includes Stations 132, 133, 134, 135, 136, 137, 138, 139, 140, and 141.

[3] Statistically significant at the $\leq$1% level, when each test (taxa) considered separately.

[4] When Wilcoxon run on this taxon with Stations 139 and 141 included in Group 1, the difference was significant at the 1% level.

Summary of Biological Results

The following points may be summarized from the biological analyses:

1. Mapping of particular taxa suggests that they may have higher abundances in close proximity to and within the original sampling grid (i.e., the dredge material disposal site).

2. Cluster analysis using the abundances for the individual station samples and for mean station abundances both suggest that stations in close proximity to the grid are more similar to each other than to more distant stations.

3. Nonparametric Wilcoxon two-sample tests revealed significant differences in abundances for nine taxa that were grouped as stations within the grid site versus more distant stations. All but one of the taxa tested exhibited greater abundances at the disposal site.

4. Mapping of physical and chemical parameters showed a similar central tendency and therefore may be interrelated.

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CONTAMINANT MOBILITY IN DIKED CONTAINMENT AREAS

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ABSTRACT

Nine dredged material land containment areas, located at upland, lowland, and island sites, were monitored during hydraulic dredging operations in fresh and brackish-water riverine, lake, and estuarine environments. Influent-effluent sampling at the diked disposal areas showed that, with proper retention of suspended solids, most chemical constituents could be removed to near background water levels. Most heavy metals, oil and grease, chlorinated pesticides, and PCBs were almost totally associated with solids in both the influent and effluent samples. The parameters which appear to have the greatest potential impact as a result of land disposal of dredged material are ammonia, soluble manganese, total mercury, and dissolved oxygen; occasionally soluble iron, zinc, and copper may exceed criteria or background levels. However, none of these should present serious problems after dilution of the effluent discharge in the receiving waters. Actively growing vegetation in disposal areas appeared to be efficient in removing ammonium nitrogen to low levels and also for filtering out suspended solids. Dissolved oxygen in effluents ranged from 0.6 to 12.5 ppm. Geochemical phase partitioning of influent and effluent solids indicated that carbonate solids of several heavy metals tended to form during dredged slurry containment, promoted by high alkalinity and pH in site waters. Metal adsorption onto suspended particles (exchangeable phase) also increased slightly, along with a significant increase in cation exchange capacity of effluent solids.

INTRODUCTION

Recent legislation has given the Corps of Engineers greater regulatory jurisdiction over lands adjacent to navigable waterways, including wetlands and drainage systems from upland areas. Since upland areas along navigable waterways are becoming scarce, there is also increasing economic pressure to create new upland areas by disposing dredged materials on wetlands and marshlands. Additionally, there is increasing emphasis on the land containment of highly contaminated or toxic dredged material, instigated by growing concern about the pollution potential of open-water or near-shore disposal operations. Irrespective of any potential environmental impact created by the disposal of dredged material in aquatic systems, one must keep in mind that land disposal produces effluent and leachate discharges, which can irreversibly impact sensitive wetland or upland habitats.

There have been only limited studies concerning the pollution potential and physicochemical changes which are induced by the disposal of dredged material in land containment areas. Some research has suggested that the mobility or availability of many harmful chemical constituents in dredged material can be accentuated by changing environmental conditions (19,20). The placement of reduced subaqueous sediments on aerobic upland soils certainly should not create stable environmental conditions. However, other studies (1,2,13,17) have failed to demonstrate significant releases of contaminants in disposal area effluent discharges, with the noted exceptions of ammonia (2,7,19,20) and orthophosphate (2,17,19). Due to the paucity of information available and conflicting findings, a comprehensive field study concerning the impact of land disposal was warranted.*

SITE SELECTION

Nine different confined land disposal areas were monitored in different geographic settings, including three freshwater and six brackish water dredging locations. Descriptions of each site are given in Table 1. These sites were chosen on the basis of dredge site sediment and water variability, including suspected high concentrations of contaminants such as oil and grease, chlorinated pesticides, PCBs, nutrients, and heavy metals. Investigated dredged material characteristics that could greatly influence the mobility of contaminants include the sediment texture, oxidation-reduction status (Eh), pH, sulfide and organic matter contents, water salinity and alkalinity, and solids to water ratio of the dredged slurries. Disposal area characteristics were also considered, including the geographic location, effective size,

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*This paper summarizes data and findings presented in Technical Report D-78-24 (June 1978), U.S. Army Engineer Waterways Experiment Station, Environmental Laboratory, Vicksburg, Mississippi 39180.

TABLE 1. DESCRIPTION OF THE DREDGING AND CONFINED LAND DISPOSAL AREAS AND OPERATIONS

					Description of Disposal Area				Description of Treatment		
Site No.	Location of Dredging Site	Location of Disposal Area	Predominant Dredged Sediment	Effluent Discharge Site	Size of Diked Area acres (ha)	Size of Treatment Area acres (ha)	Configuration	Effective Length of Treatment Area yd (m)	Overland Flow Distance yd (m)	Size of Ponded Area acres (ha)	Site Vegetation
1	Sayreville, N. J. South channel of Raritan River, km 8 (brackish water)	National Lead Industries Disposal Area No. 4, adjacent to river	Dark grey silt	Raritan River (surface brackish water)	44 (17.8)	40 (16)	Roughly circular; divided into three equal rectangular compartments; sluice box discharge; vegetated	1400 (1280)	≈700 (640)	≈35 (14)	70 percent cover of half of site by common reed grass (*Phragmites communis*)
2	Houston, Tex. Houston Ship Channel at sta 1060+00 and in ship turning basin at sta 1080+ (brackish water)	East half of Clinton disposal area, about 1.5 km inland from channel	Fine reddish sand and silt, often heavily impregnated with weathered oil	Hunting Bayou-Houston Ship Channel, 3 km east of dredging site (surface fresh to brackish water)	280 (113)	225 (91)	Rectangular; large sluice box discharge	1300 (1190)	≈200 (185)	≈200 (81)	Sparse
3	Grand Haven, Mich. Grand River, main channel at sta 120+00 (freshwater)	Verplank's Coal & Dock Co., disposal area, Ferrysburg, Mich., adjacent to river	Fine sand with some oily fines	Grand River (surface freshwater)	6 (2.4)	6 (2.4)	Roughly rectangular; sluice box discharge	250 (230)	-- --	≈6 (2.4)	None
4	Wilmington, N. C. Anchorage Basin, Cape Fear River (fresh to brackish water)	Eagle Island disposal area, between Cape Fear and Brunswick Waterways	Dark grey silt and clay	Brunswick Waterway (surface fresh to brackish)	525 (212)	400 (162)	Discharge by one sluice box and two D-shaped weirs; circular; heavily vegetated	2000 (1830)	≈1800 (1650)	≈75 (30)	Approximately 80 percent cover by dead or dormant grasses and brush; *Phragmites communis* predominates
5	Richmond, Va. James River, main channel and dock area at Deepwater Terminal (freshwater)	Disposal area on east bank of James River	Coarse sand and gravel, some light brown to dark grey silt	James River (surface freshwater)	70 (28)	35 (14)	Long and narrow; divided into three equal square compartments; sluice box discharge	1100 (1010)	≈700 (640)	≈20 (8)	Approximately 20 percent low density cover by forest and dormant undergrowth
6	Lake Charles, La. Calcasieu River, main channel near northwest end of Lake Calcasieu (brackish water)	Disposal area No. 22, 16 river km south of Lake Charles, on dredged material islands between the ship channel and lake	Dark grey to reddish-brown mixed silt, clay, and fine sand; some oily sediments	Calcasieu River (surface fresh to brackish)	185 (75)	150 (61)	Roughly rectangular; pipe discharge,* and discharge over large rectangular weir**	300** (275) 1300** (1190)	≈300 (275)	≈125 (51)	Sparse; less than 10 percent cover by large bushes and dead grasses
7	Seattle, Wash. Duwamish Waterway, Slip No. 1 (brackish to marine)	Old wastewater treatment plant sludge lagoon area north of dredging site, 60 m from waterway	Black silt-clay	Duwamish Waterway (surface brackish water)	1.9 (0.77)	1.9 (0.77)	Rectangular; divided into two equal rectangular compartments, each 46 by 85 by 4.7 m, effluent pumped from second compartment	200 (185)	-- --	1.9 (0.77)	None
8	Vicksburg, Miss. Brown Lake, upper end of 9.5 ha lake (freshwater)	Adjacent to upper end of lake	Light grey silt with light brown crust	Durden Creek-Brown Lake (surface freshwater)	5 (2.0)	5 (2.0)	Rectangular; divided into two equal compartments; sluice box discharge	440 (400)	-- --	5 (2.0)	None
9	Southport, N. C. Elizabeth River, in open channel at confluence with the Cape Fear estuary and near the Coast Guard Boat Harbor (brackish to marine)	East end of Oak Island, with dikes adjacent to Intracoastal Waterway	Black silt-clay (both sampling trips)	Atlantic Intracoastal Waterway (surface brackish water)	48 (19.4)	45 (18)	Elongated; D-shaped weir discharge; heavily vegetated	1500 (1370)	≈700† (640) ≈400†† (370)	≈5† (2) ≈20†† (8)	6 ha of thick stand of trees and bushes in lower section; 6 ha of tall grass; 4 ha in southern section of virgin salt marsh vegetation

* Day 1.
** Days 2 and 3.
† Collection trip 1 (6-7 May).
†† Collection trip 2 (17-20 May).

potential slurry residence time, degree of ponding, extent of vegetation cover, and past history of each site.

TEST PROGRAM

The relationship between slurry residence time and effluent quality was evaluated by considering all of the sites but in particular the comparisons between cross-dike and final effluent samples collected concurrently at the Sayreville, New Jersey and Seattle, Washington disposal areas. The effect of increased residence time on ammonium and phosphate release, in conjunction with pH changes, was evaluated by continuous monitoring of effluents from the Vicksburg, Mississippi disposal area during and after completion of the disposal operations. The heavily vegetated containment area at Southport, North Carolina was monitored primarily to assess what influence actively growing vegetation might have on effluent quality; the influence of dormant winter vegetation in disposal areas on contaminant release was evaluated by monitoring the heavily vegetated Wilmington, North Carolina site. Salinity effects were evaluated by comparing trends at all of the disposal areas because of the wide range of salinities encountered. However, other physicochemical variables, such as those prevalent in freshwater versus marine environments (e.g., variance in sulfide levels) were always considered in context with the salinity comparisons.

The mobility and toxicity of trace metal contaminants are regulated by the chemical compounds with which they become associated. Although the number of discrete compounds is immense, the association of a contaminant with a general group of chemical complexes can be determined by subjecting the sediment to different specific chemical extractions or treatments. These will be referred to as "geochemical phase partitioning". The association of metals with "geochemical phases" such as soluble, exchangeable, acetic acid extractable (carbonate), and manganese and amorphous iron oxide (easily reducible) sediment components was determined for influent and effluent solid phase samples from Wilmington, North Carolina; Richmond, Virginia; Lake Charles, Louisiana; and Seattle, Washington. The hydrogen peroxide-oxidizable (organic-sulfide) component was also determined in the Seattle samples. The metals, which are mainly bound in very stable crystalline matrices, were included in the analysis of final total acid digests (residual phase) of samples from the four sites.

FIELD AND LABORATORY METHODS

More than 50 different physical and chemical parameters were determined in total samples, 0.45-μm filtrates, and greater than 0.45-μm suspended solids of disposal area influents and effluents, and surface background water samples. Influents were generally collected beneath the end of the dredge discharge pipe in the turbulent mixing pool; effluents were obtained either at the outfall pipe beneath the sluice or from the back side of a weir structure; surface background water samples were collected from the water body receiving the effluent discharge. Compositing of at least three subsamples was performed in most cases to obtain more representative samples. Three to four daily samples were collected from the monitoring stations at each site either consecutively or during separate trips; six samples were obtained at the vegetated Southport disposal area but these were equally divided into an initial and final set to evaluate what effects different densities of vegetation in land containment areas might have on effluent water quality.

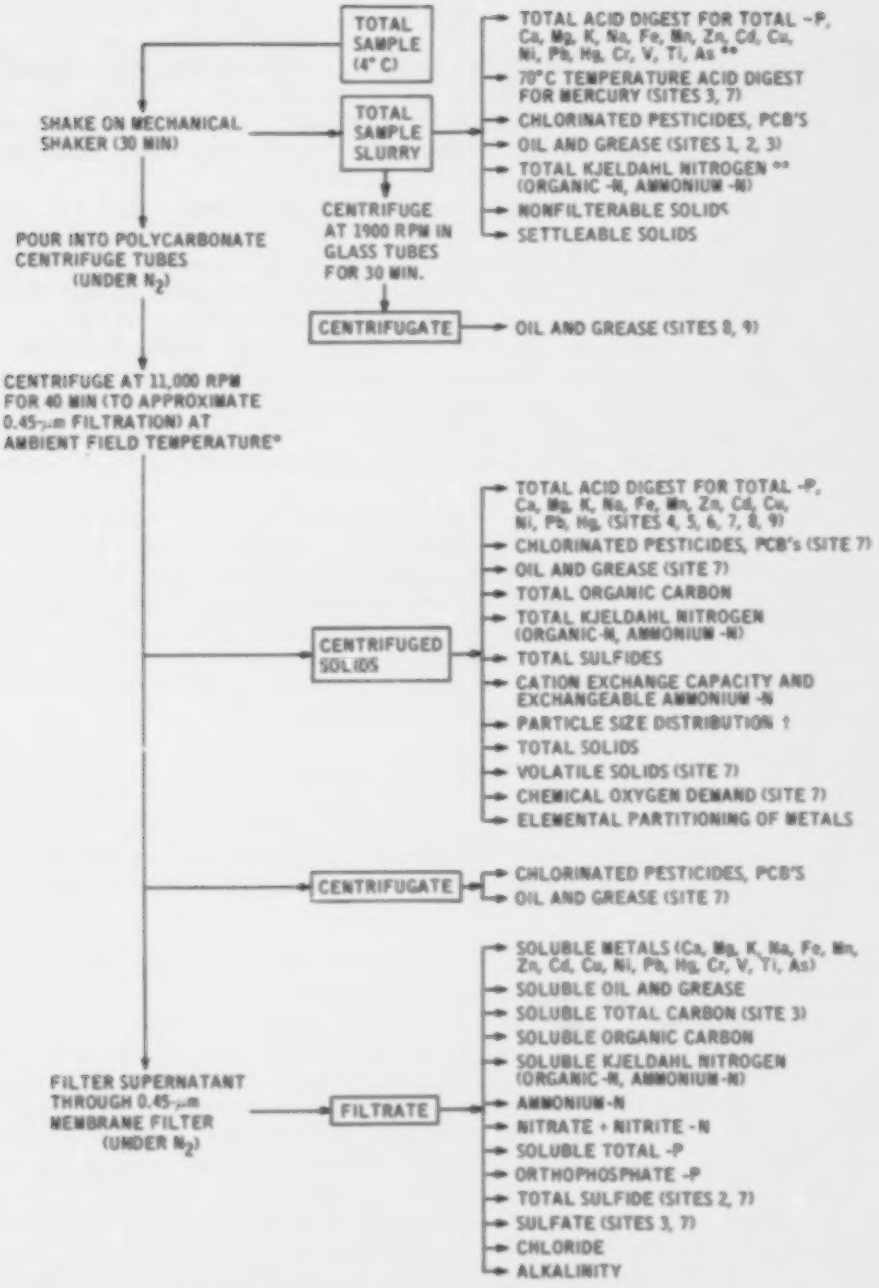
Collapsible 4ℓ polyethylene containers were employed for collection of samples used for heavy metal, nutrient, and oil and grease analyses, after being prewashed with 0.1 M hydrochloric acid and rinsed twice with deionized-distilled water. Usually, four of these containers were used; 50 ml of chloroform was added to one container as a preservative for nutrients. Samples for chlorinated hydrocarbon analyses (pesticides, PCBs) were collected in 2ℓ glass wide-mouth jars. The containers were prewashed with hexane, rinsed twice with deionized-distilled water, and combusted at 350°C for 30 minutes in a muffle furnace. All containers were completely filled with sample to exclude air, and the polyethylene containers were collapsed as aliquots were removed.

All samples were packed in ice immediately after collection and shipped by air freight to an analytical laboratory. Samples from sites 1-3 (see Table 1) were sent to the Environmental Engineering Laboratory at the University of Southern California in Los Angeles, while the samples from sites 4-6, 8, and 9 were shipped to the Environmental Laboratory at WES. Sample collection and preparation for site 7 were performed by the personnel of the Environmental Protection Agency (EPA) Region X Laboratory in Seattle, Washington. Samples were stored in environmental chambers maintained at 4°C.

Salinity, conductivity, dissolved oxygen (DO), slurry pH, and water temperature were measured in the field concurrently with influent, effluent and background water subsampling. Disposal area sediment pH and oxidation-reduction potential (Eh) were obtained in fresh sediments in the disposal areas; depending on site conditions, 6 to 40 measurements were made at each site.

Upon arrival at the analytical laboratory, the field-collected samples were processed as soon as possible to separate the solid from the liquid phase and to prepare each phase for different chemical analysis. Sample phase separation (centrifugation and filtration), total and filterable sulfides, and geochemical phase partitioning extractions were performed in a glove bag that was purged continuously with nitrogen gas to preserve the anaerobic integrity of the samples. The sequential preparation scheme for influent samples is shown in Figure 1, while the slightly different scheme used for effluent and background water samples is shown in Figure 2. The modified preparatory procedures were necessary because of the often extreme variations in solids content between respective influent, effluent, and background water samples. Generally, if an influent sample was low in suspended solids, the total sample was subjected to total acid digestions for metals, phosphorus, and total Kjeldahl nitrogen (TKN).

The parameters analyzed in the less than 0.45-mm (soluble phase) fractions of samples are listed in Figures 1 and 2. The soluble phase separations for chlorinated pesticide, PCB, and oil and grease determinations involved only highspeed centrifugation in stainless steel centrifuge tubes to approximate 0.45-μm filtration. The remaining parameters, listed in Figures 1 and 2

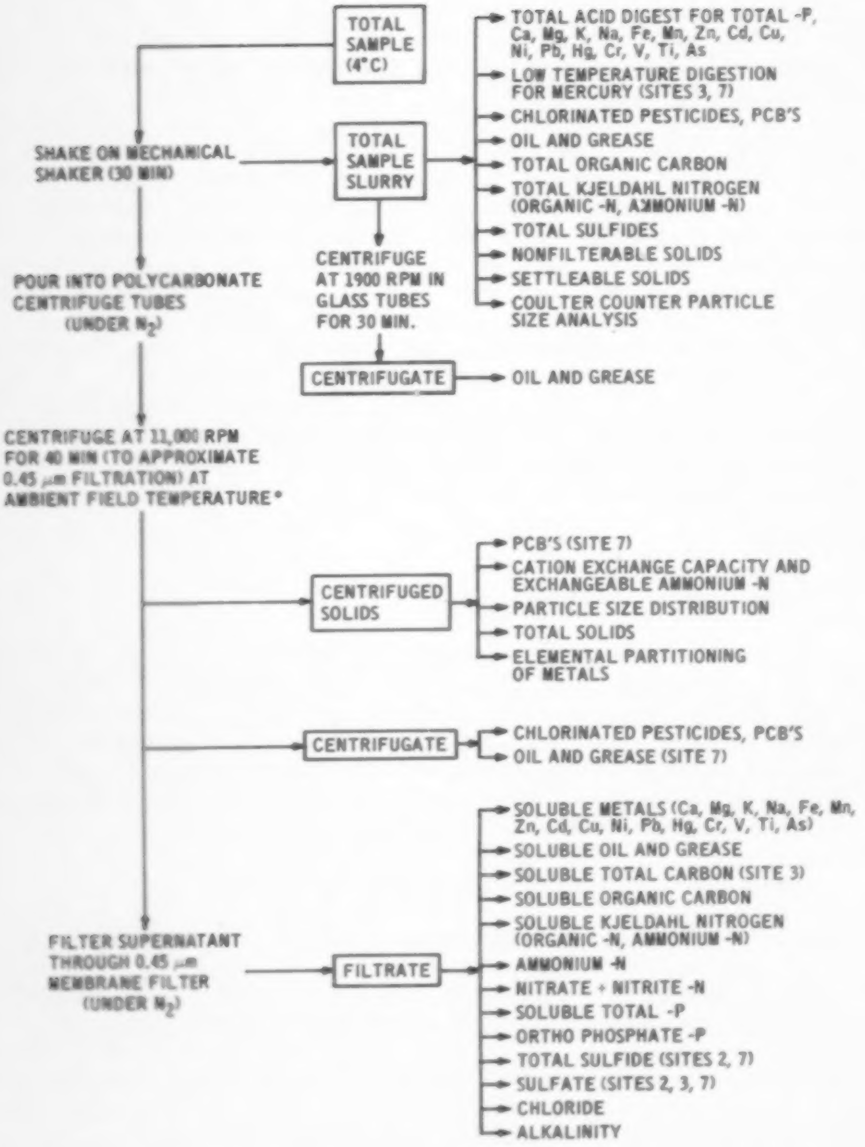

TOTAL SAMPLE (4° C)
SHAKE ON MECHANICAL SHAKER (30 MIN)
TOTAL SAMPLE SLURRY
TOTAL ACID DIGEST FOR TOTAL -P, Ca, Mg, K, Na, Fe, Mn, Zn, Cd, Cu, Ni, Pb, Hg, Cr, V, Ti, As **
70°C TEMPERATURE ACID DIGEST FOR MERCURY (SITES 3, 7)
CHLORINATED PESTICIDES, PCB'S
OIL AND GREASE (SITES 1, 2, 3)
TOTAL KJELDAHL NITROGEN ** (ORGANIC -N, AMMONIUM -N)
NONFILTERABLE SOLIDS
SETTLEABLE SOLIDS
CENTRIFUGE AT 1900 RPM IN GLASS TUBES FOR 30 MIN.
CENTRIFUGATE
OIL AND GREASE (SITES 8, 9)
POUR INTO POLYCARBONATE CENTRIFUGE TUBES (UNDER N_2)
CENTRIFUGE AT 11,000 RPM FOR 40 MIN (TO APPROXIMATE 0.45-μm FILTRATION) AT AMBIENT FIELD TEMPERATURE*
CENTRIFUGED SOLIDS
TOTAL ACID DIGEST FOR TOTAL -P, Ca, Mg, K, Na, Fe, Mn, Zn, Cd, Cu, Ni, Pb, Hg, (SITES 4, 5, 6, 7, 8, 9)
CHLORINATED PESTICIDES, PCB's (SITE 7)
OIL AND GREASE (SITE 7)
TOTAL ORGANIC CARBON
TOTAL KJELDAHL NITROGEN (ORGANIC-N, AMMONIUM -N)
TOTAL SULFIDES
CATION EXCHANGE CAPACITY AND EXCHANGEABLE AMMONIUM -N
PARTICLE SIZE DISTRIBUTION †
TOTAL SOLIDS
VOLATILE SOLIDS (SITE 7)
CHEMICAL OXYGEN DEMAND (SITE 7)
ELEMENTAL PARTITIONING OF METALS
CENTRIFUGATE
CHLORINATED PESTICIDES, PCB'S
OIL AND GREASE (SITE 7)
FILTER SUPERNATANT THROUGH 0.45-μm MEMBRANE FILTER (UNDER N_2)
FILTRATE
SOLUBLE METALS (Ca, Mg, K, Na, Fe, Mn, Zn, Cd, Cu, Ni, Pb, Hg, Cr, V, Ti, As)
SOLUBLE OIL AND GREASE
SOLUBLE TOTAL CARBON (SITE 3)
SOLUBLE ORGANIC CARBON
SOLUBLE KJELDAHL NITROGEN (ORGANIC -N, AMMONIUM -N)
AMMONIUM-N
NITRATE + NITRITE -N
SOLUBLE TOTAL -P
ORTHOPHOSPHATE -P
TOTAL SULFIDE (SITES 2, 7)
SULFATE (SITES 3, 7)
CHLORIDE
ALKALINITY

* SITE 7: 12,500 RPM FOR 20 MINUTES AT 4°C; ALL CENTRIFUGATIONS WERE MADE IN A SORVALL GSA ROTOR.

** TOTAL SAMPLE WAS DIGESTED ONLY WHEN THE SOLIDS WERE LOW.

† THE <2-μm (CLAY) FRACTION WAS FURTHER ANALYZED WITH A COULTER COUNTER.

Figure 1. Influent sample preparation.


TOTAL SAMPLE (4°C)
TOTAL ACID DIGEST FOR TOTAL -P, Ca, Mg, K, Na, Fe, Mn, Zn, Cd, Cu, Ni, Pb, Hg, Cr, V, Ti, As
LOW TEMPERATURE DIGESTION FOR MERCURY (SITES 3, 7)
CHLORINATED PESTICIDES, PCB'S
SHAKE ON MECHANICAL SHAKER (30 MIN)
TOTAL SAMPLE SLURRY
OIL AND GREASE
TOTAL ORGANIC CARBON
TOTAL KJELDAHL NITROGEN (ORGANIC -N, AMMONIUM -N)
TOTAL SULFIDES
NONFILTERABLE SOLIDS
SETTLEABLE SOLIDS
COULTER COUNTER PARTICLE SIZE ANALYSIS
POUR INTO POLYCARBONATE CENTRIFUGE TUBES (UNDER N2)
CENTRIFUGE AT 1900 RPM IN GLASS TUBES FOR 30 MIN.
CENTRIFUGATE
OIL AND GREASE
CENTRIFUGE AT 11,000 RPM FOR 40 MIN (TO APPROXIMATE 0.45 μm FILTRATION) AT AMBIENT FIELD TEMPERATURE *
CENTRIFUGED SOLIDS
PCB'S (SITE 7)
CATION EXCHANGE CAPACITY AND EXCHANGEABLE AMMONIUM -N
PARTICLE SIZE DISTRIBUTION
TOTAL SOLIDS
ELEMENTAL PARTITIONING OF METALS
CENTRIFUGATE
CHLORINATED PESTICIDES, PCB'S
OIL AND GREASE (SITE 7)
FILTER SUPERNATANT THROUGH 0.45 μm MEMBRANE FILTER (UNDER N2)
FILTRATE
SOLUBLE METALS (Ca, Mg, K, Na, Fe, Mn, Zn, Cd, Cu, Ni, Pb, Hg, Cr, V, Ti, As)
SOLUBLE OIL AND GREASE
SOLUBLE TOTAL CARBON (SITE 3)
SOLUBLE ORGANIC CARBON
SOLUBLE KJELDAHL NITROGEN (ORGANIC -N, AMMONIUM -N)
AMMONIUM -N
NITRATE + NITRITE -N
SOLUBLE TOTAL -P
ORTHO PHOSPHATE -P
TOTAL SULFIDE (SITES 2, 7)
SULFATE (SITES 2, 3, 7)
CHLORIDE
ALKALINITY

* SITE 7: 12,500 RPM FOR 20 MINUTES AT 4°C; ALL CENTRIFUGATIONS WERE MADE IN A SORVALL GSA ROTOR.

Figure 2. Effluent and background water sample preparation.

as filtrates, were determined in liquid which was both centrifuged (in polycarbonate centrifuge tubes) and then filtered through 0.45-µm nitrocellulose membrane filters. The filters were previously washed twice with 1 M hydrochloric acid and deionized-distilled water to ensure the removal of acid-leachable chemical constituents present or in the filters. Meticulous cleaning of all labware was routinely practiced, following precautions described elsewhere (6,7). All centrifugation and filtration steps were performed in a nitrogen gas atmosphere.

Following the separation of the solid and soluble phases, different aliquots were preserved according to standard procedures (10,11). The solid phase material was placed in small plastic specimen cups under a nitrogen gas atmosphere and tightly sealed. Soluble phase samples were placed in tightly capped polyethylene bottles. All of the prepared samples were stored at 4°C until further processing and analysis. Detailed descriptions of the analytical methods and instrumentation used in the study are cited elsewhere (5,12,15). Most parameters were measured according to Standard Methods (3,4) or procedures recommended by the EPA (11).

Quality control within and between laboratories consisted of multiple digestions and analyses, standard addition, and interlaboratory correlation of select samples. About half of the samples were subjected to these control measures.

Geochemical Phase Partitioning Analysis

The geochemical phase partitioning analyses were performed for influent and effluent slurry solids from sites 4-7, according to the methods outlined by Chen *et al*. (7). They include sequential extractions with chemicals which selectively remove metal contaminants associated with certain major geochemical complexes or phases. A detailed description of the procedures is cited elsewhere (12) but a generalized pairing of the extracted phases with the extractants, in sequential order, is as follows:

a. Exchangeable phase: 1.0 M ammonium acetate.

b. Acetic acid extractable (carbonate) phase: 1.0 M acetic acid.

c. Easily reducible (manganese-amorphous iron oxide) phase: 0.1 M hydroxylamine hydrochloride in 0.01 M in nitric acid.

d. Organic and sulfide phase: 30% acidified hydrogen peroxide.

e. Remaining (residual) metals): total hot acid digest.

Extractions were conducted by shaking the extractant with wet solids for 30 minutes on a mechanical shaker, followed by centrifugation and 0.45-µm membrane filtration under nitrogen gas.

RESULTS

Site Variability

The monitoring of nine different land containment areas during dredged material disposal operations was intended to aid in an assessment of the environmental impact of this mode of disposal. The general site descriptions are given in Table 1. The values for many physicochemical parameters measured in waters and sediments at each site are listed in Table 2. Many of these parameters govern the mobility of contaminants in sediment-water systems as well as being contaminants in their own right. Data on each measured parameter, including the total number of samples, ranges, means, standard deviations, and analysis of variance F values for surface background water, influent, and effluent, are presented in Table 3. The F values show whether variations between data sets are significantly different from variations within each tripartite set. The nature of the variances can be roughly assessed by consulting the means, ranges, and standard deviations given for each sample source. However, wherever variations between two means within a data set appeared to be significant, a Student's t-test was used to determine their significance.

Generally, 77 to 98% by weight of the hydraulically dredged disposal area influents consisted of sediment pore and bottom water from the dredged channel (Table 2), and at least 80% of this total water should be bottom water. Site salinities ranged from fresh to over 20 ^{o}/oo, particle size distributions from 5 to 65% sand and 11 to 65% clay-sized particles, and cation exchange capacity from 10 to 80 meq/100g of dry weight dredged sediment. Although DO was usually near zero when measured directly at the influent discharge pipe, it averaged 3.8 ppm when monitored in the mixing pool beneath the pipe discharge (Table 3).

In order to determine dredged material contaminant mobility and fate in land disposal areas, contaminant levels in influent and effluent samples were compared. Comparisons were also made between effluents and background water to roughly assess the potential impact of the effluents on water quality near the discharge site. The information in Table 3 presents a general idea of the changes in contaminant levels which were encountered during influent, effluent, and background water monitoring.

Influent-Effluent Comparisons

Comparisons between total influent and effluent digests showed prominent net decreases for all nutrients, oil and grease, PCBs, DDT, the DDT transformation product - DDD, and most major elements and trace metals during land containment as shown in Figures 3 and 4. However, site variability was great for most contaminant levels in effluents, as shown in Figure 5. Despite the often very great variance in the solids content of the influent samples, most total contaminants showed highly significant differences between influent, effluent, and background water concentrations (Table 3). The statistical non-significance shown for solid phase sulfide, DDT, DDD, and possibly total mercury is due mainly to influent solids variability, which in reality has a minimal impact on the effluents because of attenuation of the rapidly fluctu-

TABLE 2. AVERAGE VALUES FOR PHYSICOCHEMICAL PARAMETERS OF INFLUENTS, EFFLUENTS, AND BACKGROUND WATER FROM THE SAMPLED CONFINED LAND DISPOSAL AREAS

Location	Water Temperature °C	Salinity °/oo	Conductivity mmho/cm	Dissolved O_2, mg/ℓ	pH	Mechanical Particle Size, percent			Coulter Counter Particle Size, µm		Nonfilterable Solids percent Weight	Settleable Solids ml/ℓ	Cation Exchange Capacity meq/100 g	Alkalinity mg/ℓ
						Clay (<2 µm)	Silt (2-50 µm)	Sand (>50 µm)	>50 percent	>80 percent				
Sayreville, N. J.														
Influent	--	12.0	15.3	--	--	--	--	--	--	--	--	--	--	680
Effluent	17.5	14.0	17.6	6.3	--	--	--	--	--	--	--	--	--	840
Background	--	9.0	10.5	--	--	--	--	--	--	--	--	--	--	500
Houston, Tex.														
Influent	21.0	7.95	--	--	7.1	--	--	--	--	--	--	--	--	526
Effluent	19.5	8.9	--	7.85	7.7	--	--	--	--	--	--	--	--	517
Background	--	1.7	3.25	--	6.9	--	--	--	--	--	--	--	--	92
Grand Haven, Mich.														
Influent	1.5	0.7	0.52	--	--	--	--	--	--	--	--	--	--	260
Effluent	0.75	0.6	0.41	12.5	7.5	--	--	--	--	--	--	--	--	218
Background	3.5	--	0.39	11.5	7.6	--	--	--	--	--	--	--	--	140
Wilmington, N. C.														
Influent	8.5	2.8	3.2	2.3	6.6	60	31	9	1.05	0.6	7.19	467	80.6	345.1
Effluent	9.9	2.6	4.05	2.2	7.4	61	35	4	2.0	1.2	0.901	417	120.6	480.3
Background	--	--	--	--	5.5	--	--	--	--	--	0.00448	<0.1	--	17.3
Richmond, Va.														
Influent	10.6	0.25	0.29	7.75	6.7	11	24	65	1.15	0.63	1.66	85	26.2	82.3
Effluent	9.8	1.25	1.61	8.6	7.2	--	--	--	3.75	1.9	0.0464	0.7	65.9	60.5
Background	--	0	0.18	--	6.7	--	--	--	7.6	3.8	0.0010	<0.1	--	40.8
Lake Charles, La.														
Influent	14.0	15.9	20.8	4.9	7.15	64.5	29.5	6	0.91	0.56	6.26	367	56.8	320.8
Effluent	17.3	18.6	25.3	3.7	7.25	58	30	12	3.1	1.65	1.218	120	66.2	199.2
Background	--	4.2	6.8	--	6.3	--	--	--	5.4	2.82	0.0036	<0.1	--	29.9
Seattle, Wash.														
Influent	--	--	--	--	--	--	--	--	--	--	--	350	70.0	348
Effluent, Pond 1	7.1	18.5	30.1	5.4	5.8	--	--	--	--	--	0.02	0.6	--	183
Effluent, Pond 2	11.2	23.0	36.2	6.5	7.7	--	--	--	--	--	<0.01	0.25	88.0	199
Background	7.8	8.0	14.1	8.3	7.3	--	--	--	--	--	--	--	--	--
Vicksburg, Miss.														
Influent	18.5	0.15	0.65	3.0	7.0	22.5	72.5	5	0.89	0.56	23.30	950	10.3	285.5
Effluent	26.1	0.1	0.70	2.4	7.7	--	--	--	8.9	3.4	0.248	13.5	--	237.3
Background	--	--	--	--	--	--	--	--	5.5	2.85	0.00204	<0.1	--	290.0
Southport, N. C. (1)														
Influent	21.2	20.0	30.0	2.75	7.9	49	26	25	--	--	18.50	500	55.0	512
Effluent	24.0	21.3	32.0	3.3	7.65	--	--	--	--	--	0.0152	<0.1	--	178
Background	--	21.0	31.5	--	--	--	--	--	--	--	0.0094	<0.1	--	101
Southport, N. C. (2)														
Influent	22.4	15.2	23.7	2.8	7.55	41.5	50	8.5	--	--	17.10	693	57.9	1024
Effluent	30.4	20.9	38.05	6.1	7.9	--	--	--	--	--	0.1715	1.9	--	631
Background	--	--	--	--	--	--	--	--	--	--	0.02697	<0.1	--	112

Table 3

Statistical Character of Background Water, Influent, and Effluent Samples From the Confined Disposal Areas

	Number of Samples			Range			Mean			Standard Deviation			
Parameter	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Probability F Value Exceeded
Water Temp, °C	1	24	26	3.5-3.5	1.0-23.7	0.0-34.0	3.5	14.9	16.8	0.0	7.299	10.16	0.308
Salinity, 0/00	4	23	23	0.0-21.0	0.0-21.5	0.0-22.0	6.7	8.2	9.6	10.50	7.800	9.202	0.862
Conductivity, mmhos/cm	5	21	22	0.39-31.5	0.0-32.1	0.0-39.2	8.4	11.3	14.0	14.25	12.39	15.62	0.788
Dissolved O_2, mg/ℓ	1	17	23	11.5-11.5	0.25-9.3	0.6-12.5	11.5	3.8	5.3	0.0	2.885	3.402	0.048*
Slurry pH	5	16	16	5.5-7.6	6.25-8.3	6.9-8.1	6.6	7.16	7.51	0.775	0.516	0.316	0.0025**
Particle size, Clay, %	0	17	2	- -	4 -68	58 -61	- -	43.2	59.5	- -	20.15	2.121	0.280
Silt, %	0	17	2	- -	17 -78	30 -35	- -	38.1	32.5	- -	16.68	3.536	0.649
Sand, %	0	16	2	- -	0.0-79.0	4.0-12.0	- -	19.9	8.0	- -	22.50	5.657	0.479
Coulter Counter >50%, μm	3	9	10	5.4-7.6	0.77-1.2	0.2-10.5	6.2	0.97	4.75	1.242	0.131	3.545	0.0035**
>80%, μm	3	9	10	2.82-3.8	0.5-0.65	0.56-3.8	3.15	0.58	2.22	0.557	0.043	1.171	0.0001**
Total solids, % wt.	3	12	17	0.009-0.043	1.94-32.0	0.005-2.85	0.022	8.64	0.345	0.018	8.416	0.785	0.0006**

(Continued)

* Influent, effluent, and background water values are significantly different at $p \leq 0.05$.
** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

(Sheet 1 of 8)

Table 3 (Continued)

	Number of Samples			Range			Mean			Standard Deviation			Probability
Parameter	Back-ground Water	Influent	Effluent	Background Water	Influent	Effluent	Back-ground Water	Influent	Effluent	Back-ground Water	Influent	Effluent	F Value Exceeded
Nonfilterable solids, % wt.	8	17	24	0.001-0.027	0.58-32.0	0.004-3.27	0.007	11.70	0.329	0.009	10.61	0.760	0.0000**
Settleable solids ml/ℓ	7	22	24	0.1-0.1	45 -999	0.1-950	0.1	452	69	0.0	288.8	205.6	0.0000**
Total C, <0.45 μm mg/ℓ	1	3	3	58 -58	30 -85	60 -70	58	57	65	0.0	27.34	5.000	0.872
Organic C, Total, mg/ℓ	13	16	13	2 -14	35.0-11500	4 -1060	6.5	3880	151	4.521	3405	303.1	0.0000**
Solid, mg/kg	1	20	7	10 -10	974 -53400	11 -53200	10	25100	20800	0.0	16026	23677	0.385
<0.45 μm, mg/ℓ	10	29	35	4 -43	3 -185	3 - 120	15	27	19	12.89	43.32	22.31	0.456
Oil & Grease, Total, mg/ℓ	11	12	18	0.1-47.2	25.0-1497	2.4-196	6.5	458	27.5	14.32	433.0	50.99	0.0000**
Solid, mg/kg	0	5	0	- -	2928 -8492	- -	- -	6060.	- -	- -	2121	- -	- -
<0.45 μm, mg/ℓ	2	11	9	1.1-1.1	1.8-48.0	0.32-13.0	1.1	11.2	3.1	0.0	16.88	3.997	0.300
Total Chlorinated Pesticides													
op'DDE, mg/ℓ	5	14	13	<0.01-0.28	<0.01-0.53	<0.01-0.50	0.08	0.13	0.07	0.117	0.139	0.136	0.555
pp'DDE, mg/ℓ	5	14	13	<0.01-1.57	<0.01-1.72	<0.01-2.87	0.37	0.47	0.37	0.679	0.527	0.773	0.915

* Influent, effluent and background water values are significantly different at $p \leq 0.05$.
** Influent, effluent and background water values are significantly different at $p \leq 0.01$.

(Sheet 2 of 8)

Table 3 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	
op'DDD, mg/ℓ	1	14	13	<0.01–<0.01	<0.01-1.28	<0.01–<0.01	<0.01	0.19	<0.01	0.0	0.368	0.0	0.183
pp'DDD, mg/ℓ	0	14	13	- -	<0.01-1.04	<0.01–<0.01	- -	0.21	<0.01	- -	0.368	0.0	0.050*
op'DDT, mg/ℓ	1	14	13	<0.01–<0.01	<0.01-5.4	<0.01-0.23	<0.01	1.1	0.02	0.0	1.993	0.061	0.152
pp'DDT, mg/ℓ	1	14	13	<0.01-<0.0	<0.01-5.96	<0.01-0.96	<0.01	0.68	0.07	0.0	1.627	0.263	0.416
Total PCB, mg/ℓ	1	20	23	<0.01-0.3	<0.1-21	<0.1-7.66	0.058	5.81	0.50	0.090	5.451	1.607	0.0000**
Organic N, Total, mg/ℓ	17	28	26	<0.01-2.35	3.6-839	0.1-74.5	0.376	168	9.7	0.556	205.8	18.017	0.0000**
Solid, mg/kg	0	21	3	- -	532 -3870	906 -3042	- -	1820	2220	- -	1087	1150	0.562
<0.45 µm, mg/ℓ	10	28	34	0.1-1.35	0.1-27.6	0.1-6.7	0.55	4.3	1.6	0.397	5.871	1.627	0.0083**
NH_3-N, Total, mg/ℓ	16	7	21	0.01-1.56	7.3-86.0	0.82-80.3	0.53	45.6	19.6	0.459	28.46	22.93	0.0000**
Exch, mg/kg	0	18	7	- -	2.4-339	58.5-458	- -	110	196	- -	94.12	167.8	0.115
<0.45 µm, mg/ℓ	10	29	35	0.01-0.82	0.66-71.7	0.74-70.9	0.27	20.8	13.6	0.270	19.03	15.73	0.0034**
NO_3-NO_2-N, mg/ℓ	16	26	32	0.01-1.98	0.01-0.82	0.01-1.83	0.46	0.18	0.35	0.523	0.234	0.396	0.066
Total P, Total, mg/ℓ	17	28	[illegible]	0.07-0.86	12.8-496	0.11-82.1	0.26	155	11.7	0.236	133.6	21.85	0.0000**
Solid, mg/kg	0	21	3	- -	639 -4400	1400 -4000	- -	1850	3070	- -	1179	1447	0.117

(Continued)

* Influent, effluent and background water values are significantly different at $p \leq 0.05$.
** Influent, effluent and background water values are significantly different at $p \leq 0.01$.

(Sheet 3 of 8)

Table 3 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability
	Back-ground Water	Influent	Effluent	Background Water	Influent	Effluent	Back-ground Water	Influent	Effluent	Back-ground Water	Influent	Effluent	F Value Exceeded
<0.45 μm, mg/ℓ	10	29	32	0.02-0.5	0.03-9.47	0.01-1.53	0.18	0.86	0.33	0.175	2.047	0.499	0.221
Orthophosphate P,													
<0.45 μm, mg/ℓ	8	22	28	0.03-0.16	0.04-5.89	0.01-1.04	0.08	0.79	0.18	0.051	1.585	0.294	0.074
Alkalinity, mg/ℓ as CaCO$_3$	10	29	33	10.87-820	31.30-1320	29.75-670	66.10	412	287	93.08	217.4	184.5	0.0015**
Chloride, mg/ℓ	10	29	34	-20600	5.0-19200	5.0-16400	4150	8290	7810	7909	7326	6365	0.261
Total Sulfide, Solids, mg/kg	0	21	3	- -	17.8-3090	94.1-327	- -	493	208	- -	679.5	116.6	0.484
Cation Exch. Cap, meq./100g	0	19	6	- -	2.37-88.2	65.9-120.6	- -	50.9	82.5	- -	25.16	21.55	0.011*
Calcium, Total, mg/ℓ	10	24	25	4.8-390	45.7-11500	16.8-560	98.7	2450	250	142.3	3420	163.6	0.0013**
Solids, mg/kg	0	22	7	- -	1150 -37900	1190-26100	- -	16300	9930	- -	13722	10844	0.274
<0.45 μm, mg/ℓ	10	24	25	4.1-390	8.0-416	13.0-532	95.4	181.1	204.8	141.6	140.7	166.3	0.165
Magnesium, Total, mg/ℓ	10	24	25	2.5-1300	26.5-1320	3.15-1200	279.0	1270	464	514.4	1057	416.8	0.0003*
Solids, mg/kg	0	22	8	- -	933 -37800	4700-16300	- -	11200	10200	- -	8239	4853	0.746
<0.45 μm, mg/ℓ	10	24	25	1.5-1300	2.6-1300	2.6-1200	265	415	398	497.1	432.8	447.9	0.661

(Continued)

* Influent, effluent, and background water values are significantly different at $p \leq 0.05$.
** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

(Sheet 4 of 8)

Table 3 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Back-ground Water	Influent	Effluent	Background Water	Influent	Effluent	Back-ground Water	Influent	Effluent	Back-ground Water	Influent	Effluent	
Potassium,													
Total, mg/ℓ	10	24	25	2.2-338	128 -6360	4.6-1145	121	1770	390	148.3	1644	362.3	0.0000**
Solids, mg/kg	0	20	8	- -	3100 -43500	8330-18200	- -	13200	14200	- -	8655	3647	0.762
<0.45 μm, mg/ℓ	10	24	25	1.2-380	1.6-450	1.5-440	98.6	163	166	151.6	137.0	144.9	0.418
Sodium,													
Total, mg/ℓ	8	19	23	6.5-9800	85.0-9900	6.5-11300	2470	3900	3540	4026	3422	3963	0.772
Solids, mg/kg	0	11	5	- -	2394 -13100	240-43200	- -	9430	23600	- -	3583	18176	0.021*
<0.45 μm, mg/ℓ	9	24	25	6.1-9700	6.2-9500	6.0-9400	2183	3620	3490	3737	3806	3798	0.607
Iron,													
Total, mg/ℓ	17	29	34	0.38-63.6	46.1-12600	1.14-1290	5.19	3400	193	15.13	3565.	343.0	0.0000**
Solids, mg/kg	0	22	8	- -	24300-81600	24100-48300	- -	42300	38300	- -	13117	7068	0.421
<0.45 μm, mg/ℓ	10	29	35	0.001-1.43	0.043-15.9	0.01-10.1	0.378	3.52	0.814	0.457	5.009	1.816	0.0037**
Manganese,													
Total, mg/ℓ	17	29	35	0.04-0.40	0.8-310	0.21-48.5	0.107	63.1	7.9	0.095	73.00	13.61	0.0000**
Solids, mg/kg	0	22	8	- -	250-2110	683-2170	- -	682	1160	- -	383.9	470.9	0.0081**
<0.45 μm, mg/ℓ	10	29	35	0.002-0.184	0.004-14.4	0.002-7.95	0.059	2.35	1.45	0.057	3.485	2.090	0.057*

(Continued)

* Influent, effluent, and background water values are significantly different at $p \leq 0.05$.
** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

(Sheet 5 of 8)

Table 3 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Back-ground Water	Influent	Effluent	Background Water	Influent	Effluent	Back-ground Water	Influent	Effluent	Back-ground Water	Influent	Effluent	
Zinc, Total, mg/ℓ	17	25	30	0.006-1.28	0.6-206	0.026-5.49	0.238	27.5	1.03	0.422	40.28	1.481	0.0003**
Solids, mg/kg	0	22	8	- -	55.8-1960	31.7-3660	- -	323	621	- -	435	1232	0.325
<0.45 μm, mg/ℓ	10	25	30	0.001-0.121	0.001-0.496	0.002-0.228	0.028	0.055	0.064	0.042	0.10	0.069	0.472
Cadmium, Total, mg/ℓ	13	26	32	<0.0002-0.01	0.002-7.17	0.001-0.37	0.003	1.39	0.051	0.003	2.036	0.102	0.0002**
Solid, mg/kg	0	17	4	- -	0.048-45.3	0.045-4.87	- -	7.1	1.62	- -	10.44	2.216	0.319
<0.45 μm, mg/ℓ	9	29	35	<0.002-0.012	0.0002-0.015	0.001-0.011	0.003	0.004	0.003	0.004	0.004	0.003	0.824
Copper, Total, mg/ℓ	15	29	35	0.003-0.16	0.1-18.2	0.02-1.59	0.038	6.09	0.28	0.042	5.327	0.414	0.0000**
Solid, mg/kg	0	22	8	- -	6.0-165	26.0-131	- -	52.2	46.3	- -	48.43	35.00	0.753
<0.45 μm, mg/ℓ	10	29	35	0.001-0.[illegible]	0.001-0.106	0.001-0.1	0.009	0.019	0.021	0.010	0.026	0.022	0.323
Nickel, Total, mg/ℓ	14	29	30	<0.01-1.5	0.21-18.2	<0.01-1.70	0.120	5.8	0.30	0.397	4.555	0.414	0.0000**
Solid, mg/kg	0	22	8	- -	15.4-124	25.3-74.6	- -	47.3	47.1	- -	26.06	15.18	0.982
<0.45 μm, mg/ℓ	10	29	30	0.003-0.036	0.003-0.047	0.002-0.043	0.011	0.014	0.012	0.013	0.013	0.013	0.820

(Continued)

** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

(Sheet 6 of 8)

Table 3 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Back-ground Water	Influent	Effluent	Background Water	Influent	Effluent	Back-ground Water	Influent	Effluent	Back-ground Water	Influent	Effluent	
Lead,													
Total, mg/ℓ	9	28	23	0.001-0.049	0.24-86.5	0.001-7.57	0.011	16.2	1.38	0.015	23.88	2.437	0.0033**
Solid, mg/kg	0	21	8	- -	5.7-327	1.0-142	- -	81.5	43.7	- -	88.65	43.58	0.262
<0.45 μm, mg/ℓ	10	24	25	<0.001-0.005	<0.001-0.012	<0.001-0.007	0.002	0.002	0.002	0.002	0.003	0.002	0.725
Mercury,													
Total, mg/ℓ	14	18	24	<0.0002-0.009	0.001-0.243	<0.0002-0.367	0.001	0.044	0.024	0.002	0.059	0.080	0.161
Solid, mg/kg	0	18	6	- -	0.07-1.66	0.08-3.2	- -	0.46	0.79	- -	0.438	1.234	0.324
<0.45 μm, mg/ℓ	8	19	28	<0.0002-0.004	0.0002-0.008	<0.0002-0.006	0.001	0.001	0.001	0.001	0.002	0.002	0.895
Chromium,													
Total, mg/ℓ	8	3	8	0.009-0.026	56.7-76.6	0.024-0.58	0.018	63.8	0.12	0.005	11.11	0.190	0.0000**
<0.45 μm, mg/ℓ	1	3	8	0.003-0.003	0.003-0.005	0.004-0.033†	0.003	0.004	0.017†	0.0	0.001	0.011	0.132
Titanium,													
Total, mg/ℓ	1	4	4	0.01-0.01	2.1-4.35	0.255-0.50	0.010	3.31	0.36	0.0	0.956	0.117	0.0017**
<0.45 μm, mg/ℓ	1	4	4	0.0001-0.0001	0.025-0.038	0.020-0.036	0.0001	0.029	0.028	0.0	0.006	0.009	0.030*

(Continued)

* Influent, effluent and background water values are significantly different at $p \leq 0.05$.

** Influent, effluent and background water values are significantly different at $p \leq 0.01$.

† High effluent concentrations resulted from inclusion of high effluent values from the Seattle, Washington site, which lacked comparable influent data.

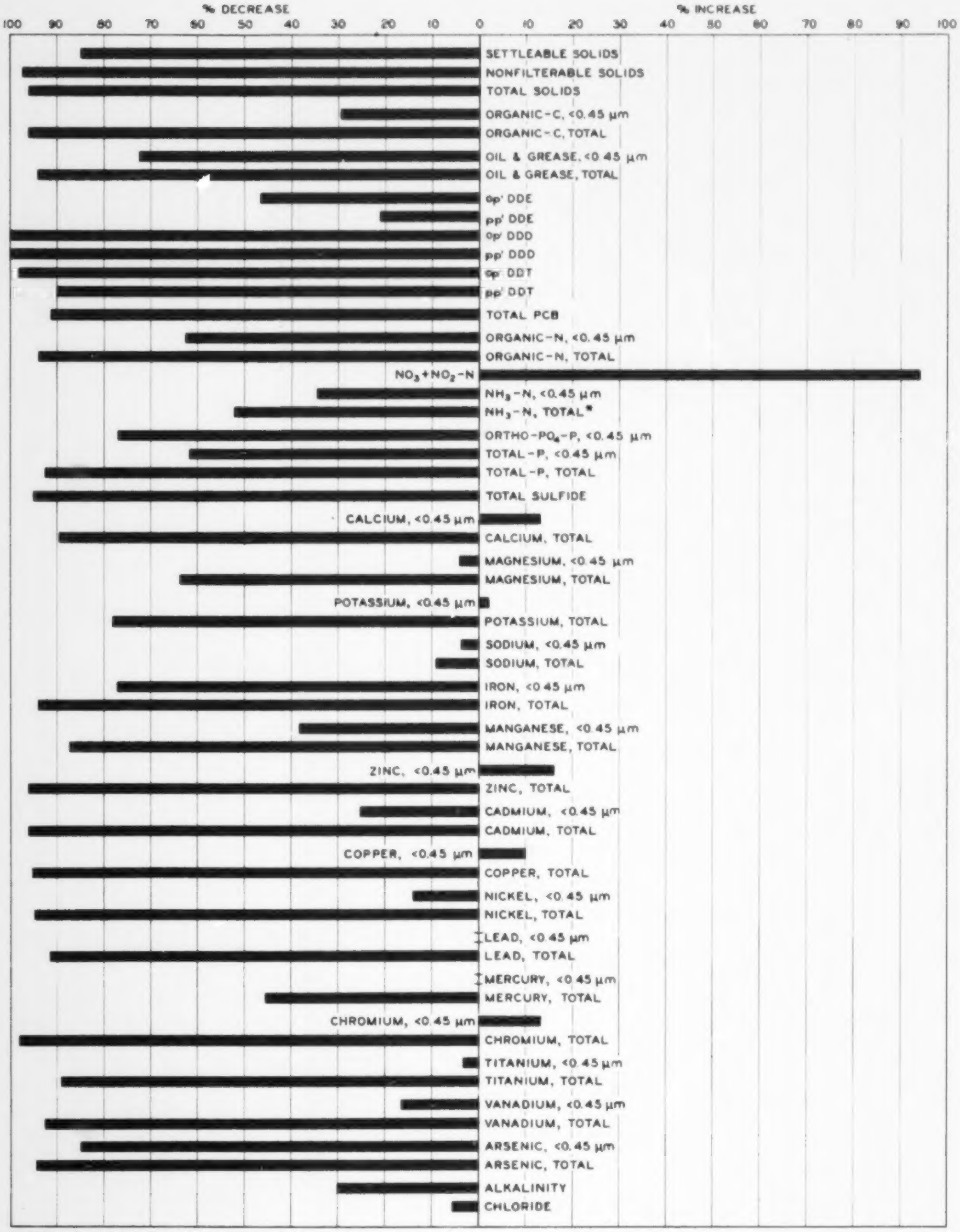
(Sheet 7 of 8)

Table 3 (Concluded)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability
	Back-ground Water	Influent	Effluent	Background Water	Influent	Effluent	Back-ground Water	Influent	Effluent	Back-ground Water	Influent	Effluent	F Value Exceeded
Vanadium													
Total, mg/ℓ	2	7	7	0.029-0.32	2.29-5.23	0.076-0.47	0.175	3.52	0.26	0.206	1.321	0.134	0.0000**
<0.45 μm, mg/ℓ	2	7	7	0.004-0.004	0.004-0.039	0.003-0.027	0.004	0.018	0.015	0.0	0.013	0.010	0.317
Arsenic, Total, mg/ℓ	9	12	17	0.001-0.013	0.181-6.02	0.003-0.41	0.004	1.73	0.096	0.004	1.979	0.126	0.0006**
<0.45 μm, mg/ℓ	2	12	17	0.0003-0.001	0.0001-0.117	0.0001-0.021	0.001	0.027	0.004	0.000	0.043	0.006	0.077

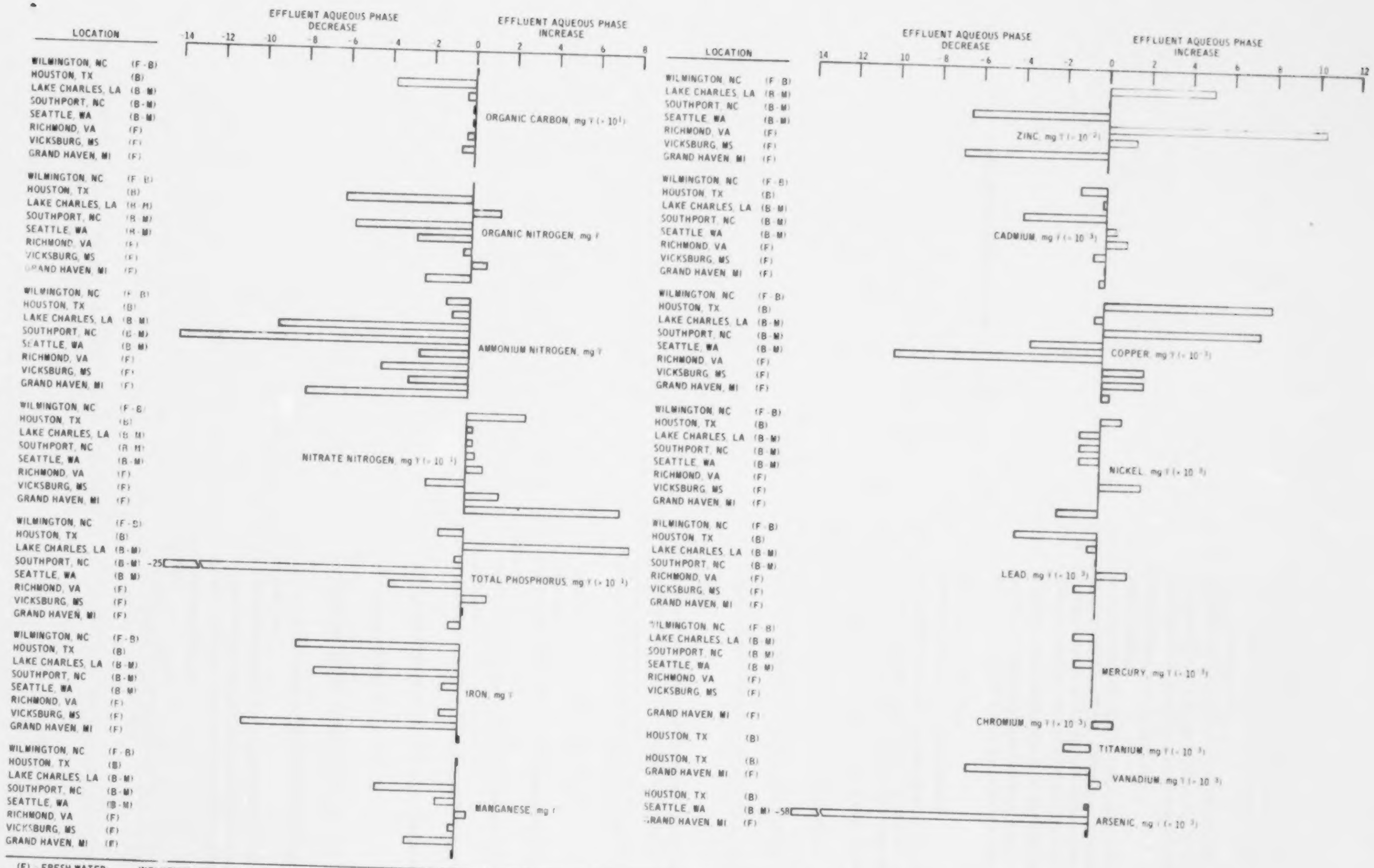
** Influent, effluent and background water values are significantly differnt at $p \leq 0.01$.

(Sheet 8 of 8)



*EXCHANGEABLE AMMONIUM-N + <0.45 µm AMMONIUM-N

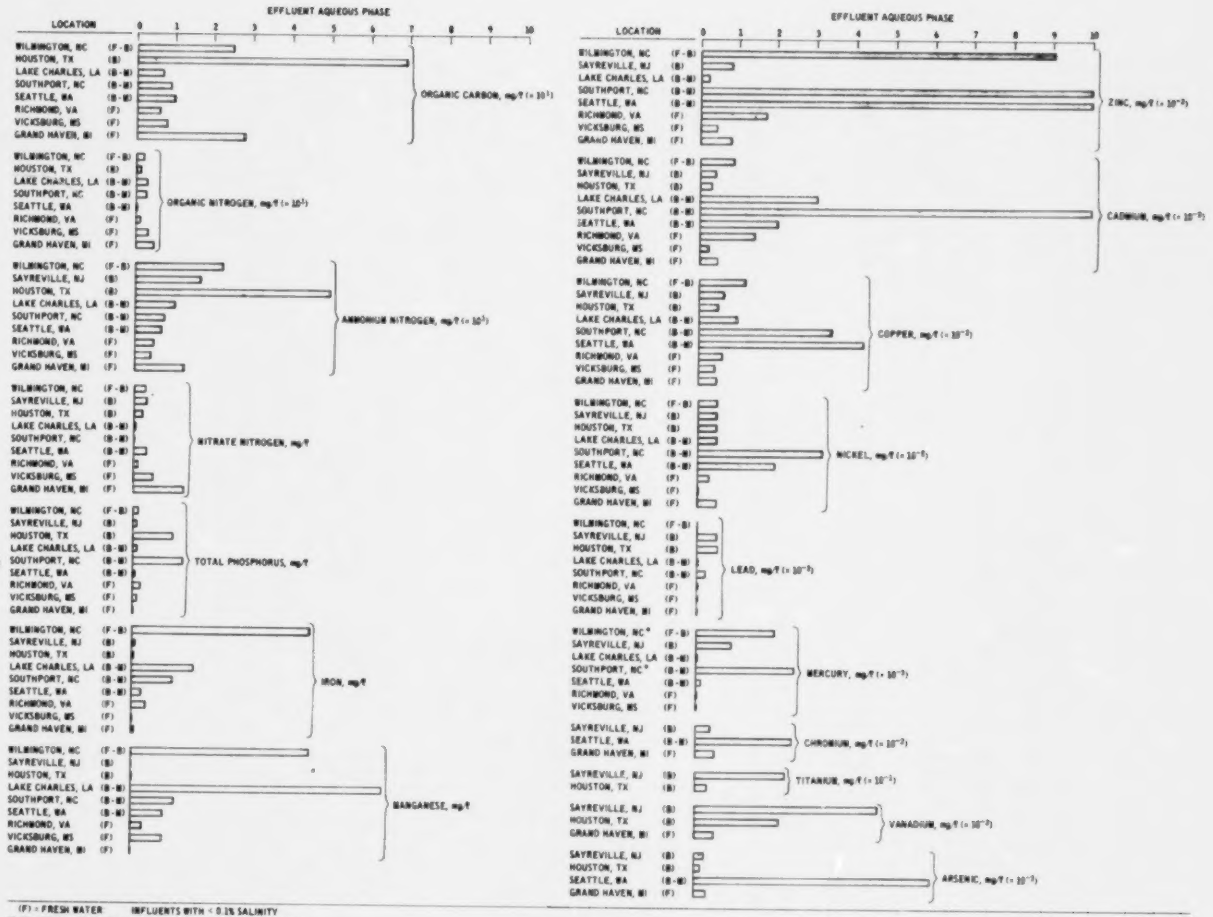
Figure 3. Percent increase or decrease of physical and chemical parameters in total and soluble phase effluent dredged material based on influent-effluent samples.



(F) = FRESH WATER: INFLUENTS WITH < 0 % SALINITY
(F-B) = FRESH BRACKISH: INFLUENTS WITH 0.1 TO 0.5% SALINITY
(B) = BRACKISH WATER: INFLUENTS WITH 0.5 TO 1.5% SALINITY
(B-M) = BRACKISH MARINE: INFLUENTS WITH > 1.5% SALINITY

NOTE: DISPOSAL AREA LOCATIONS ARE LISTED IN THE ORDER OF THEIR DECREASING DETENTION TIMES BASED ON SIZE, INFLUENT EFFLUENT FLOW RATES, AND TORTUOSITY

Figure 4. Changes in the nutrient and heavy metal concentrations in the aqueous phase of effluents from confined disposal areas.



(F) = FRESH WATER: INFLUENTS WITH < 0.1% SALINITY
(F-B) = FRESH-BRACKISH: INFLUENTS WITH 0.1 TO 0.5% SALINITY
(B) = BRACKISH WATER: INFLUENTS WITH 0.5 TO 1.5% SALINITY
(B-M) = BRACKISH-MARINE: INFLUENTS WITH > 1.5% SALINITY

* ANALYSES WERE MADE WITH A ZEEMAN SPECTROPHOTOMETER, WHICH GAVE HIGHER VALUES THAN THE COLD VAPOR METHOD.

NOTE: DISPOSAL AREA LOCATIONS ARE LISTED IN THE ORDER OF THEIR DECREASING DETENTION TIMES, BASED ON SIZE, INFLUENT-EFFLUENT FLOW RATES, AND TORTUOSITY.

Figure 5. Concentrations of nutrients and heavy metals in the aqueous phase of effluents from the confined disposal areas.

ating influents. Sodium, mercury, and the DDT transformation product - DDE were the only parameters which did not seem to be greatly reduced in the total effluents. For most elements, the percent decrease in total concentration was very close to the respective decrease in suspended (nonfilterable) solids during containment, which averaged 97.2% for all sites (Figure 3). However, some variances in elemental versus solids removal efficiency seemed to be greater than analytical error. Despite overall significant decreases in total samples, organic nitrogen, phosphorus, potassium, sodium, manganese, zinc, and mercury showed major increases in the effluent solid phase (Table 3). These contaminants were probably mainly associated with very small or low density (e.g., organic) particles. Other studies have shown similar relationships in dredged material from land containment areas (15), marine sediments (10), and in ocean discharges of wastewater effluents (8).

The only measured parameters which exhibited removal efficiencies of less than 90% in total sample preparations include: titanium (89%); manganese (88%); settleable solids (85%); potassium (78%); magnesium (64%); ammonium nitrogen (57%); mercury (46%); op´ DDE (46%); and pp´ DDE (21%). The two chlorinated pesticide analogs of DDE were at similar concentrations in both the effluent and background water samples.

Nitrate-nitrite nitrogen was the only soluble phase parameter which showed a significant increase during land containment of dredged material (Figure 3). The concentration increased almost twofold, from 0.18 to 0.35 mg/ℓ. However, this increase should have minimal impact since the discharge concentration was below the surface background water average (Table 3) and well below accepted water quality standards (9). The greater than threefold increase in soluble chromium indicated by Table 3 is the result of high levels of chromium in effluents from the Seattle disposal area; chromium was not measured in the influent samples from Seattle. Excluding the Seattle data, chromium showed only a very small increase, as shown in Figure 3. High concentrations of soluble ammonium nitrogen and manganese occurred in some influent and effluent filtrates. Soluble ammonium nitrogen in the influents averaged 20.8 mg/ℓ, with an upper range of over 70 mg/ℓ; effluent samples averaged 13.6 mg/ℓ, with maximum concentrations also around 70 mg/ℓ. Soluble manganese showed a comparable decrease during residence in the land containment areas; the influent and effluent means were 2.35 and 1.45 mg/ℓ, while maximum values were 14.4 and 8.0 mg/ℓ, respectively. Soluble iron was also high in influents but it showed a high removal efficiency of 70%. Average soluble phase concentrations of zinc and copper were slightly higher in effluents. However, as indicated in Figure 4, their increased mobility was site-specific and their levels (Figure 5) were below the present drinking water standard and criteria for aquatic life (9).

Some total alkalinity values (as calcium carbonate) were also very high, giving average influent and effluent values of 412 and 287 mg/ℓ in comparison to a background water average concentration of 88 mg/ℓ (Figure 3 and Tables 2 and 3). Dissolved oxygen in effluents, based on multiple surface water measurements made inside of the discharge weirs, fluctuated greatly. Ranges were from 0.6 to 12.5 mg/ℓ, with an average of 5.3 mg/ℓ. Thus low effluent DO can occasionally be a problem.

Geochemical Partitioning of Influent and Effluent Solids

The potential mobility or toxicity of an element is highly dependent on the geochemical phases with which it is associated in the solids fraction. The exchangeable ions are considered to be readily available to aquatic organisms as they are mainly weakly adsorbed on the surfaces of fine solids. The carbonate phase may also be readily affected by changes in the environment, especially as a result of interactions with or uptake by the biota. Also the carbonates of a given metal are generally more soluble than many other solid phase chemical precipitates, and thus a major shift of a metal to a carbonate complex may result in a similar increase in its soluble phase concentration. The easily reducible phase variations reflect the oxidized or reduced status of a containment area during disposal operations. An increase in this phase would suggest an overall increase in the oxidation of the dredged material slurry, and vice versa. If reducing conditions occur, increased mobility of ions from the easily reducible phase (iron-manganese oxide precipitates) is favored, thus presenting a potential water quality problem. In retrospect, greater mobility may occur from sulfide complexes (organic-sulfide phase) under oxidizing conditions since most metal sulfides are much less soluble than their more oxidized substituents. Thus with changing environmental conditions many contaminants change their mobility and form, and likewise their bioavailability (18).

Figure 6 gives the amounts and percentages of each of 14 nutrient and trace metals that were solubilized during sequential chemical extractions of five influent and effluent solids. Some metals exhibited noticeable phase changes during migration of suspended solids in the dredged slurry across land containment areas, while other metals showed little change. About a third of the solids-bound calcium and sodium were removed from the influent solids during extraction of the exchangeable phase, with measurable increases in exchangeable calcium, sodium, copper, and arsenic noted in effluent solids. Exchangeable phase manganese, magnesium, and cadmium were similarly high (~10%) in both influent and effluent solids. Most metals showed increases in carbonate phase concentrations as a result of confined disposal. Influent solids generally showed high carbonate phase values for cadmium and manganese, while zinc, cadmium, manganese, lead, copper, and sodium showed major increases in effluent solids. Carbonate phase cadmium, zinc, and manganese composed 57, 33, and 20% of the effluent solids, respectively. Iron, manganese, cadmium, and copper increased in the easily reducible phase of the effluent solids, although only manganese showed a major increase of 20%. Upon total digestion of the remaining influent and effluent solids, most metals (except for iron, nickel, and chromium) showed noticeable decreases in the effluent digests. A limited amount of data on the organic-sulfide phase (Seattle site) suggests that the concentration decreases were mainly associated with reductions in solid phase organic and/or sulfide complexes during disposal area detention under oxidizing conditions.

Metals showing major geochemical phase changes included manganese (easily reducible phase), cadmium (carbonate and easily reducible phases), zinc (carbonate phase), lead (carbonate phase), copper (carbonate phase), sodium (exchangeable phase), and calcium (exchangeable phase). Metals showing little change in phase during the solids detention times included chromium, nickel, mercury, potassium, and magnesium. Iron could not be properly evaluated because

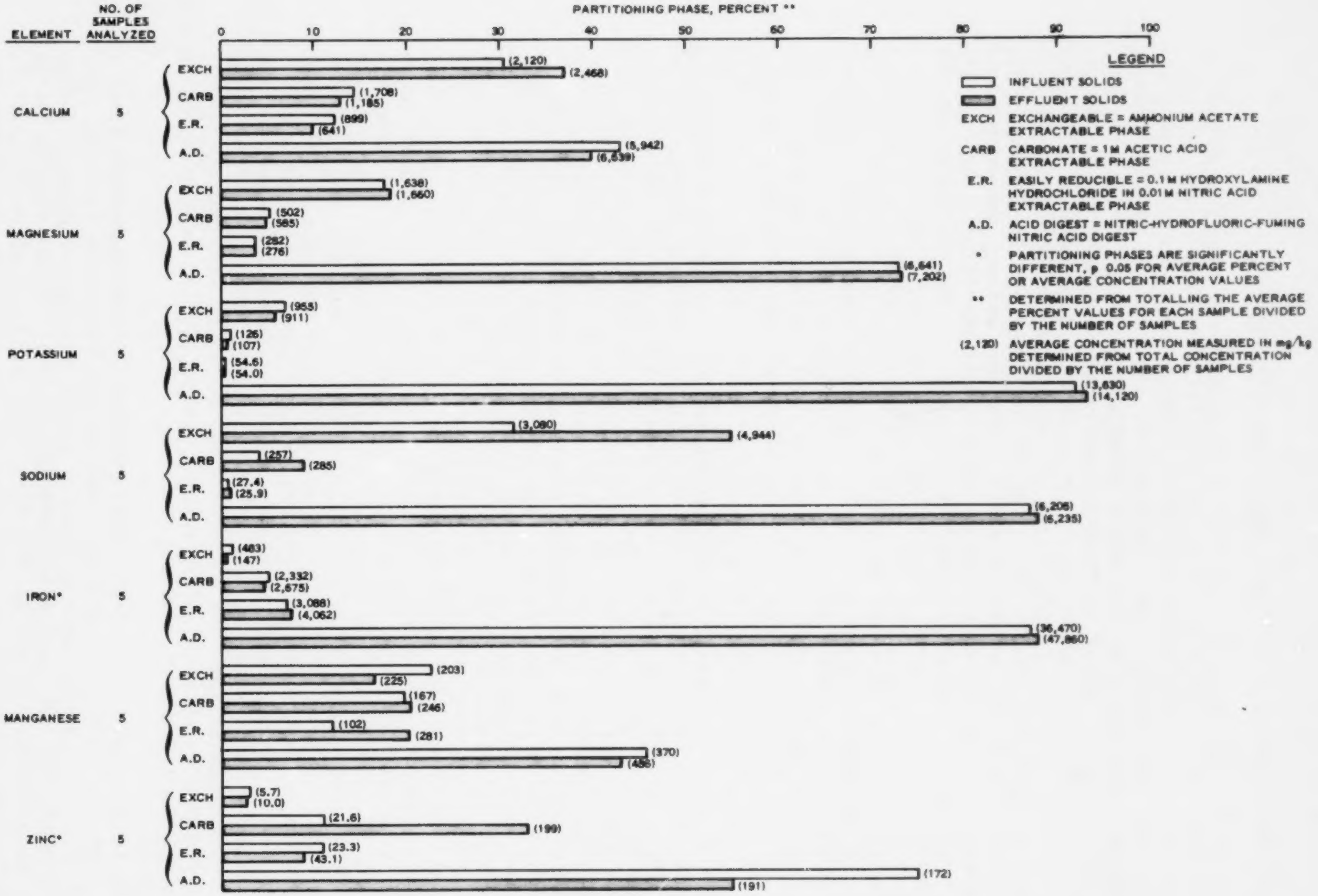


Figure 6. Geochemical phase partitioning of metals in influent and effluent solids from four confined land disposal areas (sheet 1 of 2).

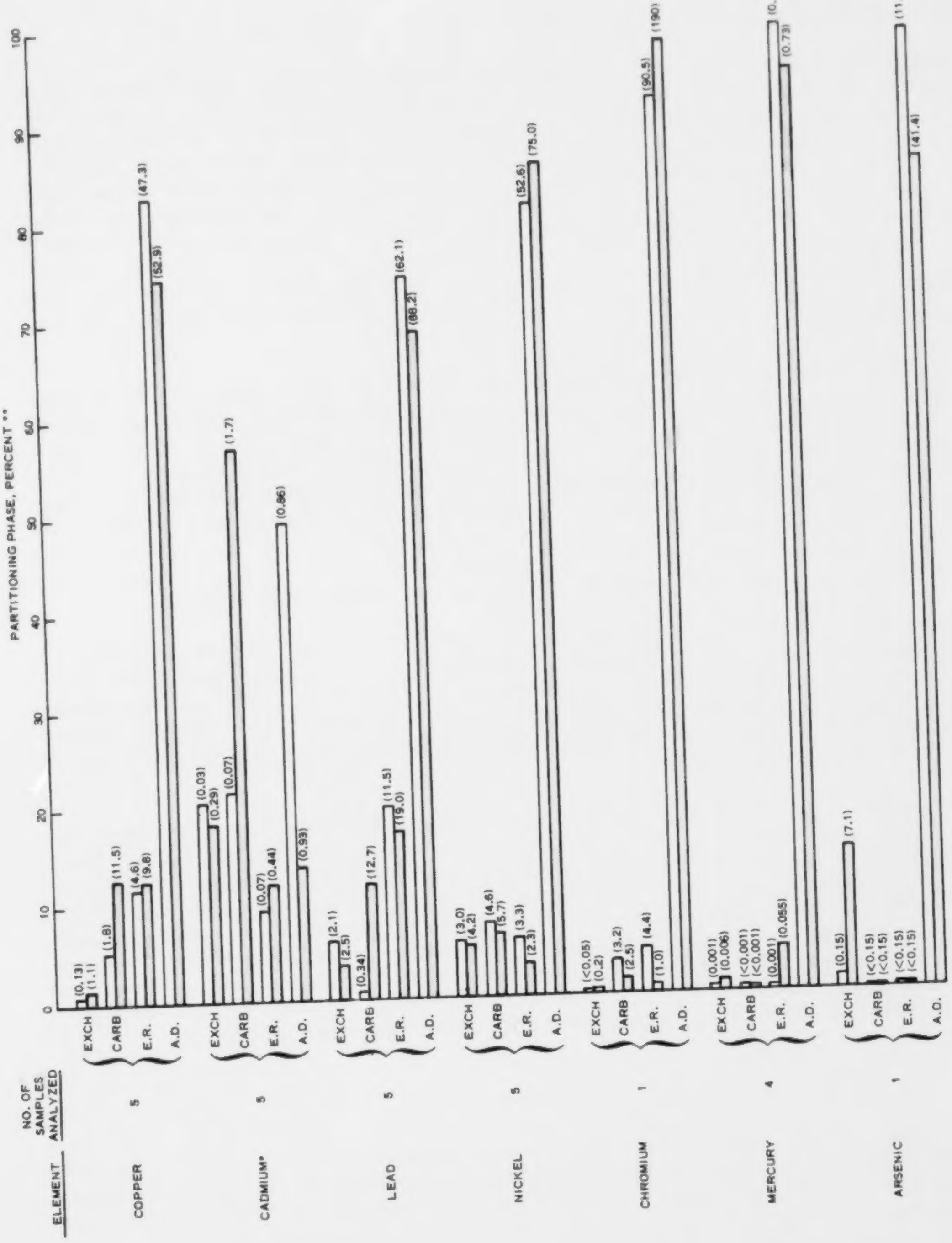


Figure 6. (sheet 2 of 2)

of its high total concentration. Arsenic showed a large increase in an exchangeable phase extract of an effluent sample from the Seattle site.

Effluent-Surface Background Water Comparisons

Suspended (nonfilterable) solids were 47 times higher in effluents than in adjacent surface background water (Table 3). Therefore, it is not surprising that most total sample contaminant levels were also correspondingly higher, as depicted in Figure 7. The only contaminants which were significantly higher in the effluents than in the background water were total lead (125X) and total manganese (74X). Contaminants which had appreciably lower levels in total effluent samples than the respective suspended solids contribution included: PCBs, DDT, DDD, DDE, copper, chromium, sulfide, zinc, nickel, vanadium, calcium, magnesium, potassium, and sodium (Table 3). The major source for the total manganese, DDE, and major ions (Ca, Mg, Na, K) was the soluble phase.

The only measured soluble phase parameters which were appreciably higher than the background water levels include: ammonium nitrogen (50X), manganese (25X), alkalinity (3X), zinc (2.5X), copper (2.5X), and iron (2.5X) (Figure 7). Titanium and vanadium were significantly higher in effluents in the Houston site, but were at background levels at two other monitored sites. Despite the data given in Figure 7, chromium and arsenic were not shown to be higher in effluents since high effluent values for each at the Seattle site could not be matched with missing background water data. More information is required for those trace contaminants which are not commonly monitored.

The primary source for the DDE and nitrate levels in effluents seemed to be water incorporated with the dredged sediments since higher levels for each were present in surface background water samples (Table 3). The soluble phase concentrations of the major ions (Ca, Mg, Na, K, Cl) were significantly higher in effluents than in background water because of salinity stratification in most brackish dredge site waters.

The association of trace contaminants with very fine particles was observed. This was particularly evident for mercury and potassium at most sites. Combined geochemical phase partitioning and particle-size fractionation of effluent solids show the association of potassium with very fine silicate mineral particles, e.g., illite clay. Fine clays, as well as manganese and iron oxide precipitates and low density organic detritus, may be important media for the transport of adsorbed contaminants. This is supported by high exchange capacity (Table 3) and increased levels of exchangeable cations (Figures 3 and 6) associated with effluent suspended solids.

DISCUSSION

The removal efficiency for most heavy metals closely paralleled the removal of the solids during dredged slurry containment in land disposal areas. However, different metals seemed to have varying affinities for different geochemical phases and particle sizes of suspended solids. Total mercury in effluent samples decreased by only 46%, which indicates that it was

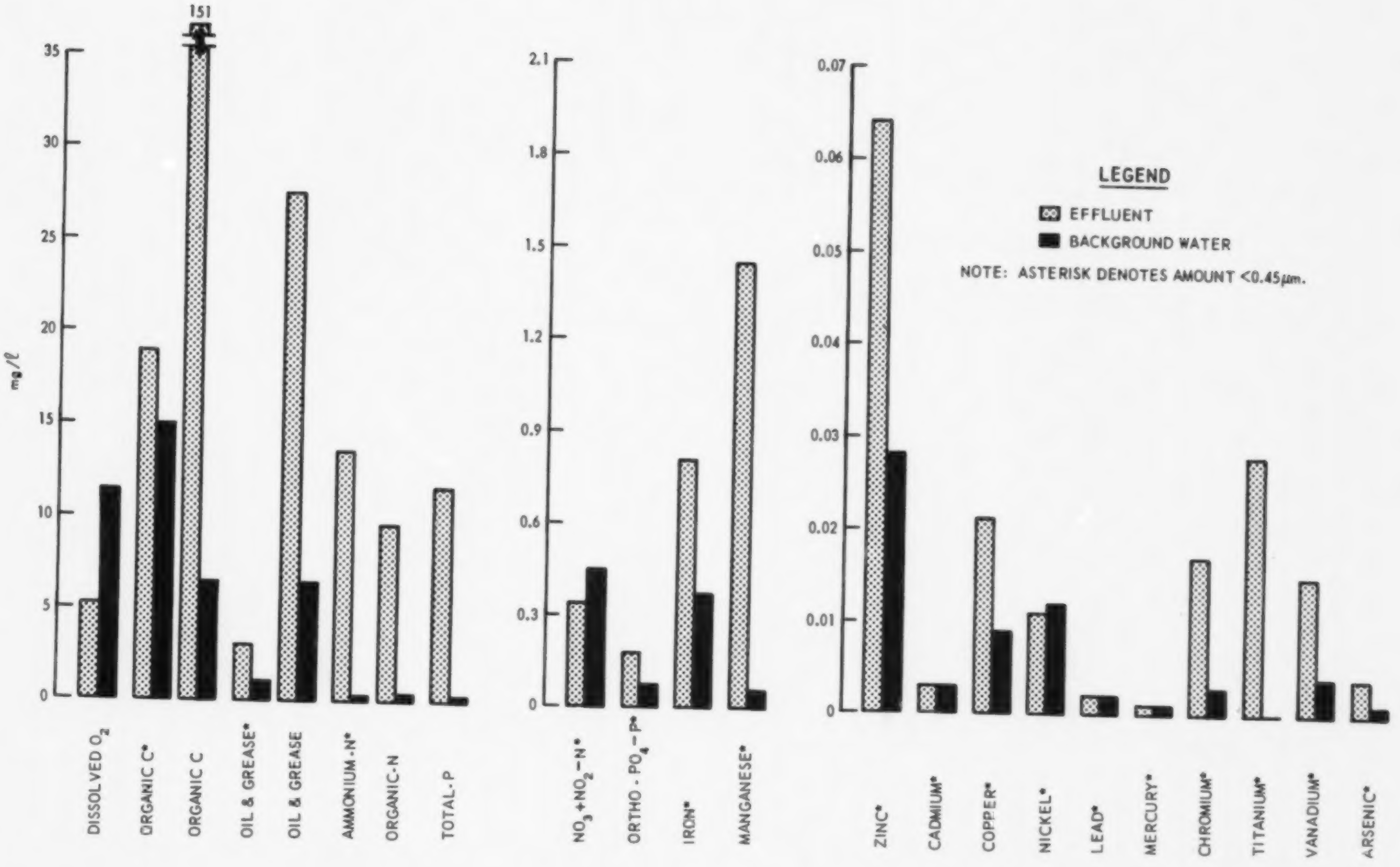


Figure 7. Average concentrations of nutrients, organic contaminants, and soluble phase heavy metals in effluents and surface background water samples representing the confined disposal areas.

often associated with a fine particulate fraction and/or one of low specific gravity (e.g., organic suspended solids). Soluble phase titanium in the Sayreville samples mainly originated from the settling of very fine titanium oxide aerosol particles into the water near the dredging site. Most of the DDE also seemed to be associated with very fine particles present in dredged site waters. Thus, the filter size and the instrument employed for analyses are of great importance in determining "soluble phase" concentrations. The impact that fine particulate matter has on aquatic life is not well documented, but the findings of this study suggest that perhaps suspended solids should be included in any predictive test or effluent analysis; the methods used to separate solids and analyze samples should be well documented. However, bulk analyses (acid digests) of bottom sediments or total influents are not recommended as a predictive tool as they generally show a poor relationship to contaminant mobility or availability.

Most of the chlorinated hydrocarbons (pesticides, PCBs) with the exception of DDE, showed very efficient removal when proper solids retention was maintained in confined disposal areas. The source for the DDE was probably not the dredged sediments since comparable DDE concentrations were observed in surface background water samples. Oil and grease were generally removed efficiently during dredged slurry containment. However, sediments containing high levels of petroleum residues seemed to settle more slowly, often creating highly fluid oil-water-sediment suspensions near the bottom of ponded areas. Poor site management may effect the release of these suspensions, resulting in a poor effluent quality.

Analytical data for influent and effluent sample filtrates showed that soluble phase ammonium nitrogen was released in high concentrations from some bottom sediments. Ammonium release was most frequently directly related to organic nitrogen and TKN concentrations in the bottom sediments. Soluble phase ammonium nitrogen concentrations in disposal area influent samples averaged 20.8 mg/ℓ with maximum levels of 70 mg/ℓ. Generally, an equivalent amount of ammonium nitrogen was exchangeable from the influent solids. A very rapid initial decrease in soluble phase ammonium was noted in most sites displaying a short slurry detention. This was attributed to sorption by disposal area solids in contact with the slurry and was most pronounced in the presence of fine-grained sediments. Although ammonium nitrogen was removed from the dredged slurry by 57% during residency, the observed effluent concentrations could warrant concern, especially if high pH conditions exist in the disposal area or discharge zone, promoting the formation of highly toxic un-ionized ammonia.

Soluble manganese and iron were the only heavy metals consistently released at above ambient background water concentrations in effluent filtrates. Soluble manganese removal in the disposal areas was generally good, averaging 38%. Soluble iron was readily removed during slurry residency. However, both iron and manganese frequently greatly exceeded present standards and criteria. Although copper and zinc concentrations were higher in the soluble phase of many effluents, the levels encountered suggest that only rarely should these contaminants exceed present standards and criteria. The increased mobility of zinc and copper during containment could be promoted by their release from and complexation by organic compounds as well as the formation of moderately soluble carbonate minerals indicated by their notable increases in the acetic acid extractable solids phase. Generally, the soluble phase concentrations of other heavy metals closely reflected the concentrations in comparable background water samples. Soluble phase cadmium showed no major change in concentration during confinement, but there was a major shift of cadmium into the carbonate phase of effluent solids. This suggests that cadmium associated with effluent suspended solids may have greater potential for bioavailability or mobility in receiving waters. However, solid phase cadmium seemed to be readily removed during short-term residency in land disposal areas.

There was a 60% increase in the cation exchange capacity of effluent solids. This increase was reflected by general increases of exchangeable ammonium and metals associated with the effluent solids. The dominance of fine particulate matter in effluents was probably responsible for this. Manganese and other metal increases in the easily reducible geochemical phase also suggest the importance of manganese and iron hydroxide precipitates in sorption phenomena. However, effluents having low solids contents should be negligibly affected by the increase in exchange capacity. Thus dredged slurry retention must be sufficient to allow for efficient suspended solids removal.

Although a direct relationship between carbonate phase shifts and influent-effluent alkalinity values was not consistently noted, high alkalinity undoubtedly played an important role. Alkalinity showed an overall decrease during containment, although the trends were site specific. Major shifts in alkalinity during dredged slurry containment seemed to be promoted by biological activity. The highest influent alkalinity values were noted when the dredged sediments contained a high total organic carbon concentration.

There was a small but significant increase in pH during the dredging and land disposal cycle (Table 3). Algal photosynthesis appeared to be an important source for the increase, with pH values in excess of 9 observed in effluents from a nonvegetated disposal area with a retention time of greater than a week. High nutrient levels can increase algal growth in site waters, resulting in alkaline pH. The combination of high nutrient and pH levels may have both positive and negative effects; ammonia volatilization may lead to nitrogen depletion of site waters and increased ammonia toxicity could impact both the microbial pathogens in the dredged material and aquatic organisms in the receiving waters. High turbidity and vegetation in the disposal areas should prevent excessively high pH.

Dissolved oxygen averaged 5.3 mg/ℓ in effluents, and generally effluents containing higher solids contents were lower in DO. Some very clear effluents which contained high concentrations of soluble nutrients were also low in DO. Subsurface effluent discharge had lower DO than surface discharge. Low DO values were also observed in vegetated overland flow disposal areas; this may be prompted by the turbulent mixing and greater contact of reduced sediments with the water in overland flow systems. Influents entering the vegetated overland flow areas were also unusually high in nutrients, which may have prompted the low levels of DO by accentuating microbial growth.

In summary, trace contaminants are mainly associated with suspended particles, and turbid water will impact sensitive aquatic life. Recently disturbed bottom sediments, resulting from dredging, will also suppress DO levels in the immediate discharge waters. Therefore good disposal area management requires removal of suspended solids. However, long-term retention of the dredged material supernatant in disposal areas could produce a high pH problem. Although slightly alkaline pH generally favors the removal of soluble contaminants (14,18), very high values are not desirable. If organic matter and TKN values are high, ammonia released with high pH effluent discharges may be toxic to sensitive organisms in the immediate receiving waters. Since the main source for the high pH is probably algal photosynthesis in disposal area water, algal blooms should be watched for and prevented. Such biological blooms could also create unstable physicochemical conditions during the rapid death phase, resulting perhaps in rapid decreases in DO, Eh, and pH. In turn, biochemical and geochemical phases of contaminants may shift, which could increase their mobility.

The increases of nutrient trace metals in soluble phase effluent samples from the Wilmington disposal area, where site water was subjected to long residency in dead vegetation, indicate that the mobility of especially zinc and copper may be favored under conditions of extensive retention time and presence of abundant organic debris (Figure 4). Other studies have shown similar trends (19,20). However, the Southport disposal area, which contained actively growing thick vegetation, elicited the best removal of nutrients (ammonium and soluble phosphate) and suspended solids of any of the sites monitored, and no significant increases in soluble trace metals were found. Thus, there seems to be a seasonal pattern for best treatment. The presence of vegetation in disposal areas can accentuate slurry solids removal, but it is recommended that such dredging be scheduled for the spring or summer period for best effluent quality. In retrospect, disposal into nonvegetated areas may be best during cooler weather, to avoid algal blooms.

Proper planning and management of land disposal area operations should be a goal of dredging concerns and regulatory agencies. Based on present trends in criteria and standards, the main goal for achieving acceptable effluent water quality should be the removal of suspended solids, even at the expense of increasing the concentrations of certain soluble phase trace contaminants. Most soluble phase trace contaminant levels in effluents were found to be within present water quality guidelines, except for ammonia, manganese, and iron, and longer residency under oxidizing conditions should favor their removal. However, it must be kept in mind that there are seldom close relationships between total contaminant levels and toxicity or biological accumulation (16,18-20). Certainly much more research must be performed on such relationships before a clear understanding of impact of confined land disposal of dredged material can be formulated.

CONCLUSIONS AND RECOMMENDATIONS

1. Most contaminants (chlorinated hydrocarbons, oil and grease, nutrients, and trace metals) are primarily associated with the solid phase of dredged material. Ammonium, manganese, and possibly iron, copper, and zinc are the only contaminants which might occasionally exceed both background water levels and present water quality standards and criteria following effi-

cient suspended solids removal Other soluble phase contaminants in effluents were either below presently prescribed criteria or comparable to background water concentrations. Thus proper retention or treatment of site waters to limit suspended solids levels seems necessary to meet the present regulatory guidelines. However, background water levels should also be considered in properly evaluating and regulating excessive effluent levels.

2. Major shifts of several trace metals from organic-sulfide and residual to carbonate and exchangeable geochemical phases were noted during residency of dredged material solids in land containment areas. Despite a potential increase in mobility of most metal carbonate complexes, increased adsorption onto and coprecipitation with fine effluent solids probably prevented notable soluble phase increases. Only zinc and copper were noted to increase in effluent filtrates at several sites, but levels were below present regulatory guidelines.

3. Actively growing vegetation in land disposal areas elicited excellent removal of nutrients (ammonium and soluble phosphate) and suspended solids. However, seasonal use of such sites during active plant growth is suggested for best effluent quality.

4. Effluent monitoring and predictive testing methodologies should include suspended solids. Chemical data for total settleable solids or bulk sediments should not be used to indicate environmental impact.

5. Confined land disposal of dredged material, under proper management, seems to be a viable method for the containment and treatment of most contaminated sediments. However, sites located adjacent to sensitive wetlands in confined water bodies are not recommended without extensive predictive testing.

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MATHEMATICAL MODEL OF PHOSPHORUS RELEASE FROM LAKE SEDIMENT

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ABSTRACT

This paper reports the results of a study on the phosphorus release mechanism of lake sediment, primarily by examining the relationship between release rates and sediment depth. Sediment samples from Lake Nakanoumi were used to examine the relationship between phosphorus content and sediment depth and to measure the released quantities of nutrients, specifically phosphorus. From these data, a model of phosphorus release rates was conceived and evaluated. This study concludes that phosphorus release rates from sediment depend primarily on phosphorus content of the sediment.

INTRODUCTION

Lake Nakanoumi is situated in the western district of mainland Japan, which faces the Nippon Sea (Figure 1). The lake has a surface area of 96.9 km^2, a water volume of 521 x 10^8 m^3, a length of 81 km and a mean depth of 5.4 m. It joins Lake Shinjiko at its inlet and flows into the Nippon Sea. Two major cities, Matsue and Yonago, face the lake and others, Izumo and Sakai are located nearby (Figure 2). Most of the lakes adjacent to urban areas in Japan are extensively polluted because of industrial development and recreational pressures. This lake is no exception and the back portion of Yonago Bay is highly eutrophic.

The Izumo Construction Bureau, a branch of Japan's Construction Ministry, began dealing with this pollution problem several years ago; recently a plan was formed for lake restoration by sediment removal. As part of the project, the Bureau requested the Japan Bottom Sediments Management Association to perform tests on sites proposed for dredging and spill water treatment, and develop predictions for lake water quality after restoration. In this manner, we were able to conduct release experiments on the sediment, and then examine the effects of dredging and reduction of internal phosphorus loading. Sampling stations are shown in Figure 3. In Yonago Bay, which has a surface area of about 15 km^2, dredging had been routine to maintain shipping routes. But this is the first time dredging will be done as part of the lake restoration process.

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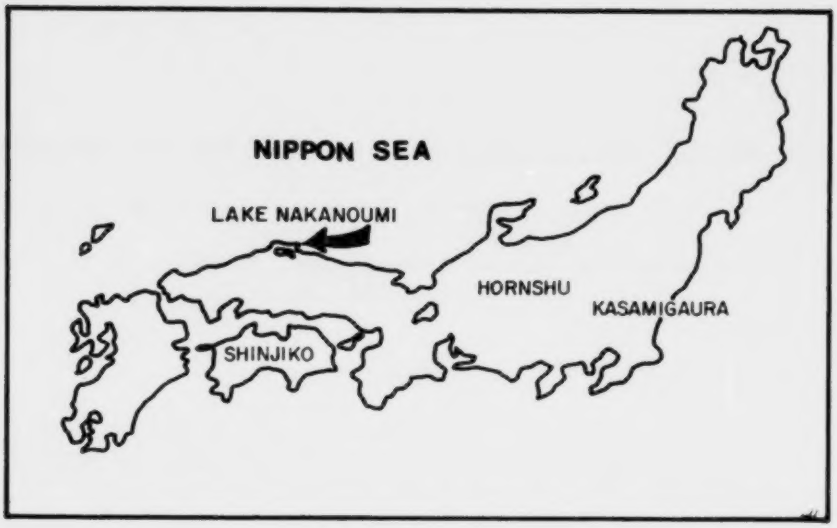


Figure 1. Location of Lake Nakanoumi.

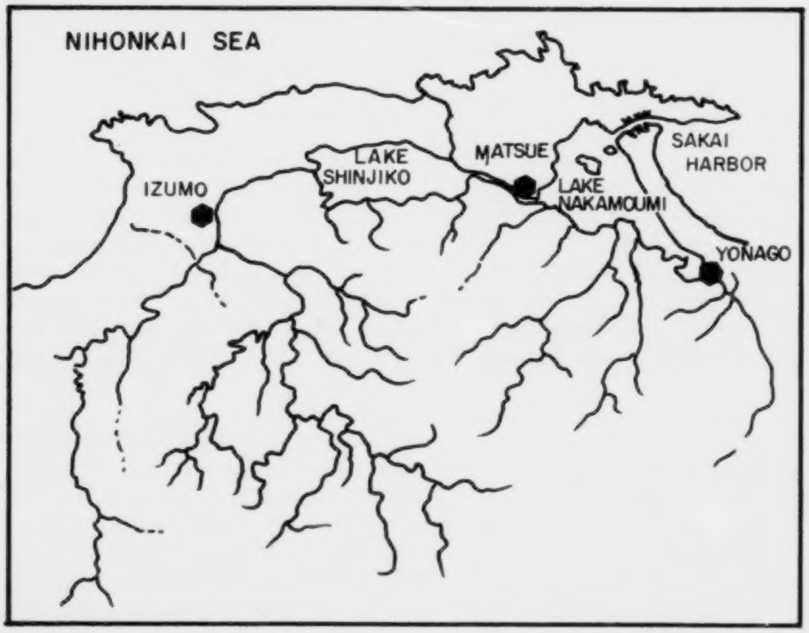


Figure 2. The river basin of Lakes Shinjiko and Nakanoumi.

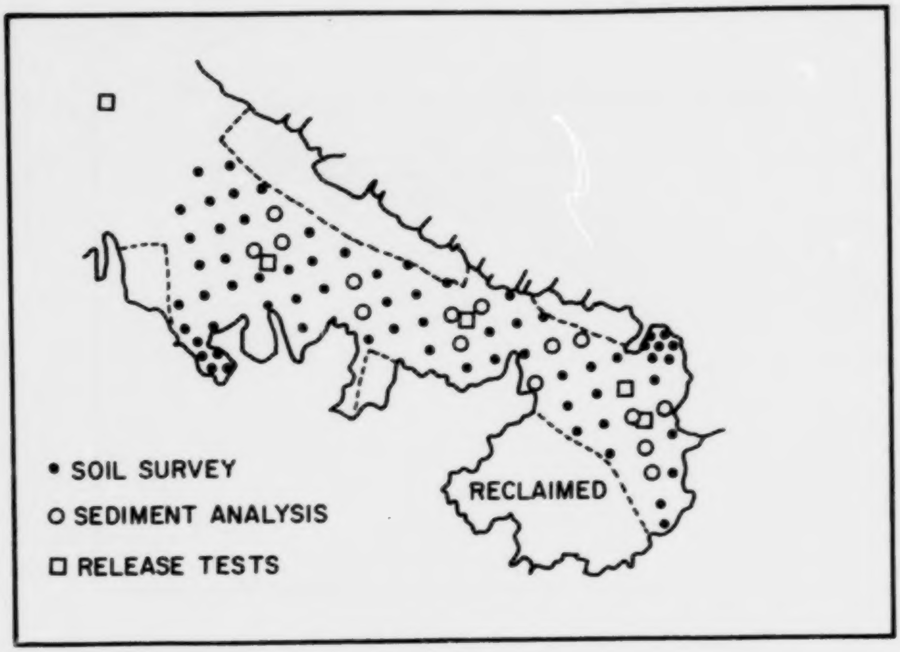


Figure 3. Sampling sites in Yonago Bay.

To prepare for the restoration plan, the following tests were performed:

-- 87 sites were observed for soil classification and soil color.

-- 16 sites provided samples for sediment analysis.

-- 5 sites were checked for phosphorus release and AAP (Algal Assay Procedure) tests.

DISTRIBUTION OF PHOSPHORUS CONTENT IN SEDIMENT

The phosphorus concentration of Nakanoumi's sediment is distributed according to sediment depth, as shown in Figure 4.

The phosphorus concentration is highest at the surface, and decreases rapidly to a depth of approximately 50 cm. Beyond a depth of 50 cm the concentration is relatively uniform. It may be assumed that the upper portion of the curve represents an accumulation of polluted sediment, and the lower portion of the curve, a background value representing an unpolluted state.

As Figure 4 shows, the sediment phosphorus distribution curve for Lake Nakanoumi is parabolic. Data from other lakes confirm this pattern, for example, the distribution curve for Lake Koyamaike is shown in Figure 5.

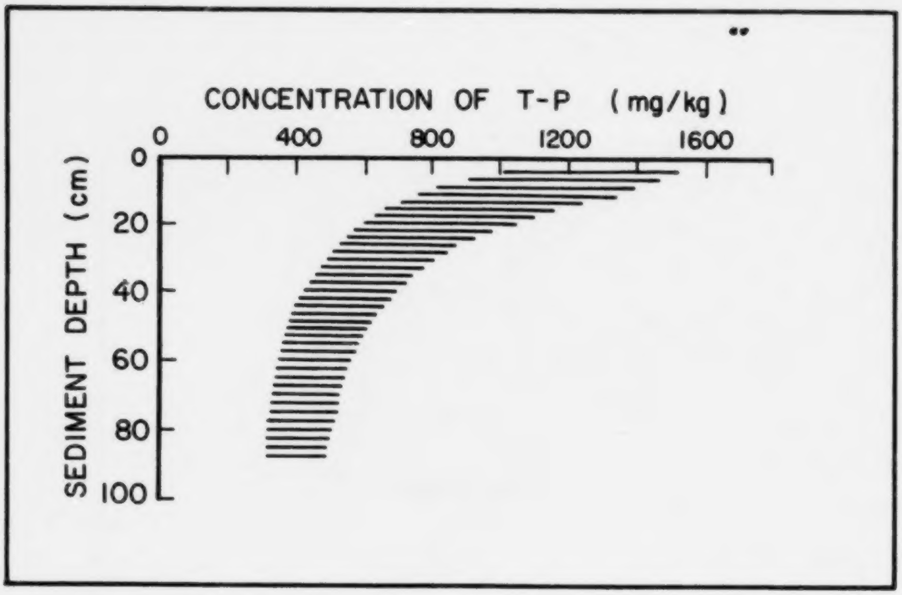


Figure 4. Distribution of phosphorus concentration with depth in Yonago Bay, Lake Nakanoumi, Japan.

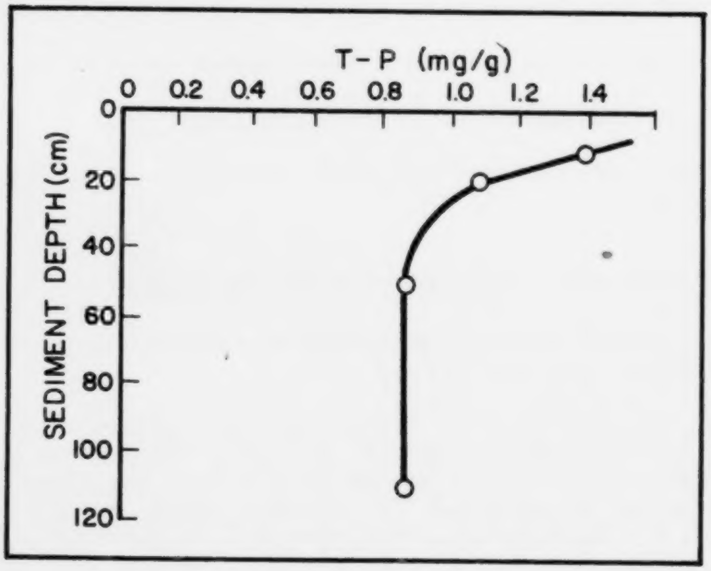


Figure 5. Distribution of T-P concentrations in Lake Koyamaike sediment.

Assume that the phosphorus concentration is represented by the following equation:

$$P_s = P_\beta + \frac{(Z_\beta - Z)^2}{2f_p} \qquad (1)$$

where:

P_s = Phosphorus in sediment (mg/Kg)

P_β = Background value of phosphorus concentration

Z_β = Sediment depth at P_β

f_p = Figure index of parabolic curve (see Figures 6 and 7)

Z = Sediment depth

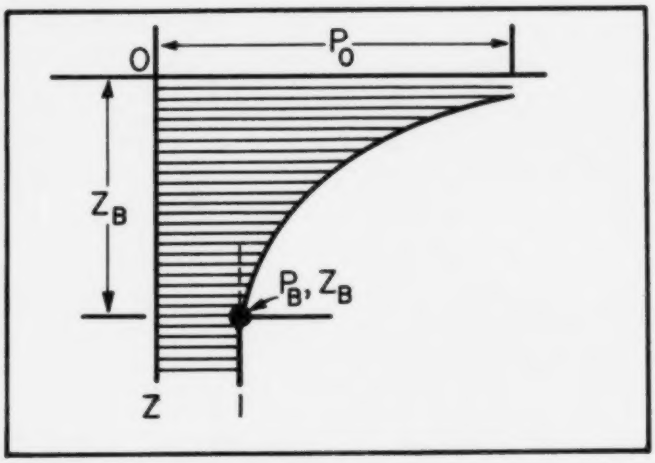


Figure 6. Illustration of parabolic curve for Equation (1) variables.

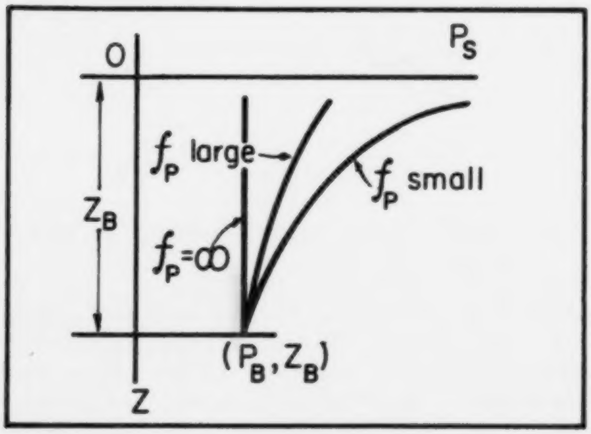


Figure 7. Figure index f_p of parabola for Equation (1).

The more horizontal the parabola, the smaller the value of f_p, and the more vertical the parabola, the larger the value; $f_p = \infty$ is represented by a vertical line. The mean concentration at sediment depth Z (mg/kg) becomes:

$$\bar{P}_s = \frac{1}{Z}\int_o^Z \left\{ P_\beta + \frac{(Z_\beta - Z)^2}{2f_p} \right\} dZ = P_\beta + \frac{Z_\beta^3 - (Z_\beta - Z)^3}{6f_p Z} \qquad (2)$$

The dry weight (kg) of the sediment layer for thickness Z is

$$\frac{\gamma_s}{1 + e_o} F \cdot Z$$

where:

F = area of column, cm^2

e_o = void ratio of sediment

γ_s = unit weight of dry sediment (usually 0.0026 kg/cm^3)

Thus, in the following equation we derive the total amount of phosphorus contained in the sediment at depth Z (mg):

$$\bar{W}(Z) = \frac{0.0026 F}{1 + e_o} \left\{ P_\beta Z + \frac{Z_\beta^3 - (Z_\beta - Z)^3}{6f_p} \right\} \qquad (3)$$

DISTRIBUTION OF PHOSPHORUS CONCENTRATION IN PORE WATER

The phosphorus concentration in pore water (Figure 8) can be obtained from the following equation:

$$D \frac{\partial^2 C_p}{\partial Z^2} = 0$$

where:

C_p = phosphorus concentration in pore water

D = diffusion coefficient in the vertical direction in sediment pore water

The diversion of molecular particles in the sediment pore water is different from that in normal free water. The diffusion coefficient is inversely proportionate to the coefficient of viscosity, which is in proportion to the solid concentration of sediment. As Figure 9 shows, the water content of the sediment in Lake Nakanoumi is high at the surface, and low in the deeper layers. This is also true for other sediments. Because of differences in

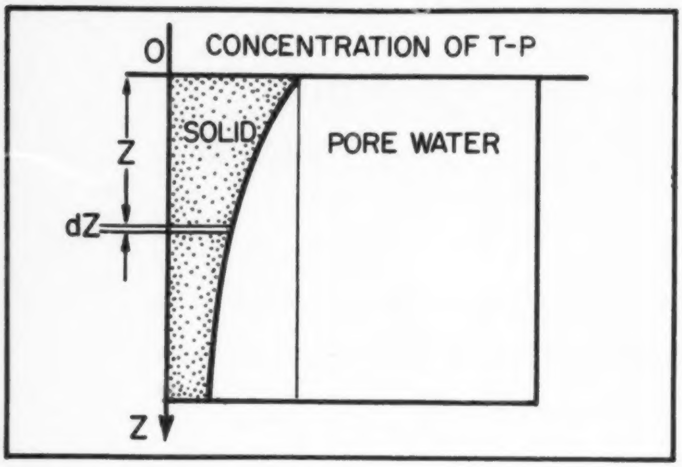


Figure 8. Concentration of pore water.

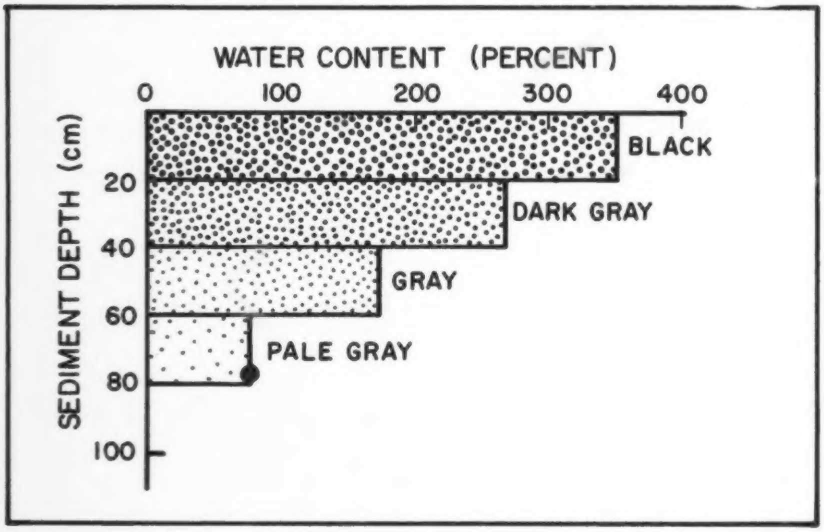


Figure 9. Relationship between water content and sediment depth in Lake Nakanoumi.

liquid viscosity, the diffusion coefficient is larger at the surface and smaller in the deeper layers. It can be assumed from this, that a reduced molecular diffusion rate in sediment indicates the sediments are from deeper strata.

Therefore, it is possible to assume that

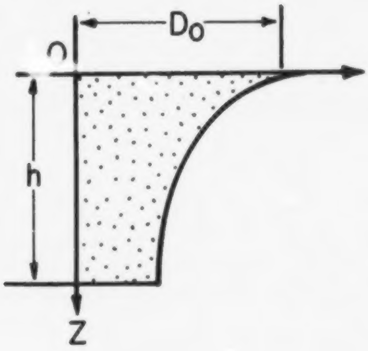
$$D = D_o - \frac{Z(2h - Z)}{2f_D} \tag{5}$$

where:

D_o = Diffusion coefficient at the surface

h = Sediment depth, where D approaches uniformity

f_D = Figure index of parabola



As previously mentioned, the Index f_D can be applied to all sediment pore water conditions. If f_D is equal to ∞, the diffusion coefficient D is constant against depth. Here Equation (4) is rewritten as

$$\left\{ D_o - \frac{Z(2h - Z)}{2f_D} \quad \frac{\partial^2 C_p}{\partial Z^2} \right\} = 0 \tag{6}$$

If we assume a solution of Equation (6) in the following series form:

$$C_p = A_o Z^m + A_1 Z^{m+1} + A_2 Z^{m+2} + \ldots \tag{7}$$

and substitute:

$$Cp'' = (m - 1)\, mA_o Z^{m-2} + m(m + 1)\, A_1 Z^{m-1} + (m + 1)(m + 2)\, A_2 Z^m + \ldots \tag{8}$$

In the given equation, we obtain

$$(m-1)m\, A_oD_oZ^{m-2} + \left\{ m(m + 1)\, A_1D_o - (m - 1)m\, A_o \frac{h}{f_D} \right\} Z^{m-1}$$

$$+ \left\{ (m + 1)(m + 2)\, A_2D_o - m(m + 1)\, A_1 \frac{h}{f_D} + (m - 1)m \frac{1}{2f_D} \right\} Z^m$$

$$+ \left\{ (m+2)(m+3)\, A_3D_o - (m+1)(m+2)\, A_2 \frac{h}{f_D} + m(m+1)\, A_1 \frac{1}{2f_D} \right\} Z^{m+1} + \ldots = 0 \quad (9)$$

To satisfy this relationship identically, it is necessary to do the following:

$$(m - 1)\, m\, A_oD_o = 0$$

$$m(m + 1)\, A_1D_o - (m - 1)\, m\, A_o \frac{h}{f_D} = 0$$

$$(m + 1)(m + 2)\, A_2D_o - m(m + 1)\, A_1 \frac{h}{f_D} + (m - 1)\, m \frac{1}{2f_D} = 0$$

$$(m + 2)(m + 3)\, A_3D_o - (m + 1)(m + 2)\, A_2 \frac{h}{f_D} + m(m + 1)\, A_1 \frac{1}{2f_D} = 0$$

Hence, we get:

$$A_1 = \frac{m - 1}{m + 1} \frac{h}{f_DD_o} A_o$$

$$A_2 = \frac{(m - 1)\, m\, (2h^2 - f_DD_o)}{2(m + 1)(m + 2)\, f_D^2D_o^2}$$

$$A_3 = \frac{(m - 1)m\, (h^2 - f_DD_o)\, h}{(m + 2)(m + 3)\, f_D^3D_o^3}$$

Thus we have:

$$C_p = A_oZ^m + \frac{(m - 1)\, h}{(m + 1)f_DD_o} A_oZ^{m+1}$$

That is, the concentration of overlying water is constant against the column height (Figure 10).

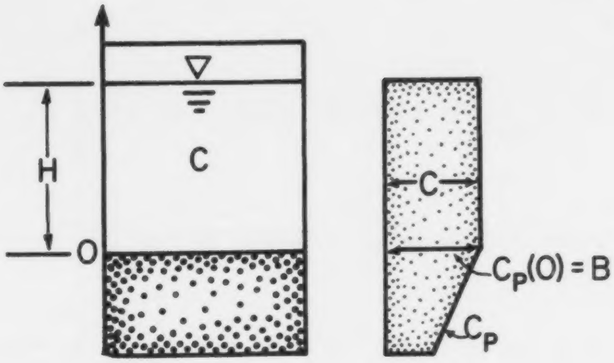


Figure 10. Concentration of overlying water.

THEORETICAL RELEASE RATES

As mentioned previously, the amount of phosphorus (mg) contained in the sediment depth Z has been determined.

$$\bar{W}(Z) = \frac{0.0026}{1 + e_o} F \left\{ P_\beta Z + \frac{Z_\beta^3 - (Z_\beta - Z)^3}{6f_p} \right\} \tag{3}$$

We now assume that the phosphorus concentration on the sediment surface increases by dc during the short time dt. The increased quantities of phosphorus in the pore water and overlying water are equal to the quantities of phosphorus particulates partitioned from the solid phase to the pore water phase. Then we have:

$$\xi \bar{W}(Z)\,dt = dC\,(H + Z)\,F \times 10^{-3} \tag{14}$$

where:

ξ = partition coefficient of phosphorus particles

Obviously, the partitioning of phosphorus in the sediment is influenced by a variety of factors; e.g., biological, chemical and hydrostatic. Therefore, the partition coefficient involves all of them (Figure 11).

$$+ \frac{(m-1)\,m\,(2h^2 - f_D D_o)}{2(m+1)(m+2)\,f_D^2 D_o^2} A_o Z^{m+2}$$

$$+ \frac{(m-1)\,m\,h\,(h^2 - f_D D_o)}{(m+2)(m+3)\,f_D^3 D_o^3} A_o Z^{m+3} + \ldots \qquad (10)$$

Since $(m - 1)\,m = 0$, provided $m = 1$ or $m = 0$

$m = 1 \;;\; C_{p1} = A_o\,Z$

$m = 0 \;;\; C_{p2} = A_o - \frac{h}{f_D D_o} A_o\,Z$

The complete solution where $A_o = 1$, is then:

$$C_p = AZ + B\left(1 - \frac{h}{f_D D_o} Z\right) = B + \left(A - \frac{h}{f_D D_o}\right) Z \qquad (11)$$

This is the distribution curve of phosphorus concentration in pore water, which is a straight line.

DISTRIBUTION OF PHOSPHORUS CONCENTRATION IN OVERLYING WATER

The distribution of phosphorus concentration in overlying water is derived from the following equation:

$$D_o \frac{\partial^2 C}{\partial Z^2} = 0 \qquad (12)$$

The solution is:

$C = -CZ + D$

The boundary conditions are:

$Z = 0 \;;\; C = C_p\,(0) = B \qquad \therefore\; D = B$

$Z = H \;;\; D_o \frac{\partial C}{\partial Z} = 0 \qquad \therefore\; C = 0$

Hence we have:

$$C = C_p\,(0) = B \qquad (13)$$

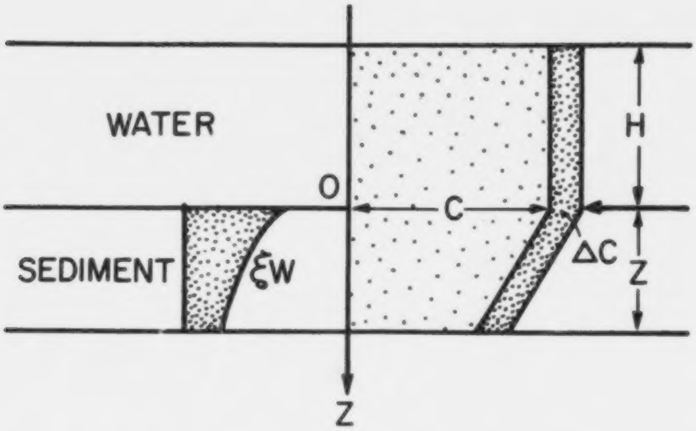


Figure 11. Partitioning of phosphorus particulates.

From equation (14), we have:

$$\frac{dC}{dt} = \frac{2.6\ \xi \left\{ P_\beta Z + \dfrac{Z_\beta{}^3 - (Z_\beta - Z)^3}{6f} \right\}}{(1 + e_o)(H + Z)} \qquad \text{as, (mg/l/day)}$$

Hence:

$$C = C_o + \frac{2.6\ \xi \left\{ P_\beta Z + \dfrac{Z_\beta{}^3 - (Z_\beta - Z)^3}{6f} \right\}}{(1 + e_o)(H + Z)} \cdot t \qquad \text{as, (mg/l)} \tag{15}$$

In release tests this concentration is measured with time. Since the expression for the phosphorus quantities which are released to the overlying water is $(C - C_o)\ F\ H \times 10^{-3}$ mg, we have the theoretical release rates as follows:

$$Pr = \frac{(C - C_o)\ F\ H \times 10^{-3}}{Ft}$$

$$= \frac{26\ \xi \left\{ P_\beta Z + \dfrac{Z_\beta{}^3 - (Z_\beta - Z)^3}{6f} \right\}}{(1 + e_o)(1 + \frac{Z}{H})} \qquad \text{as, mg/m}^2\text{/day} \tag{16}$$

RELEASE TEST RESULTS

The release tests were conducted using a columnar system with the device shown in Figure 12. Sediment samplers are made of 15 cm x 70 cm acrylic pipes. Samples were taken from undisturbed sediments in thicknesses of 7.5 cm, 11.5 cm and 30 cm.

The overlying water was collected with the sediment samples. The samples were taken so that the overlying water was protected from the air by a paraffin film when small quantities were removed for analysis. The test results are summarized in Figure 13. From these tests and analyses we can identify a relationship between phosphorus release rates and depth of sediment (Figure 14). The curve in Figure 14 indicates that the phosphorus release rates increase significantly in proportion to sediment depth for shallow cores, but levels off rapidly as core depth increases.

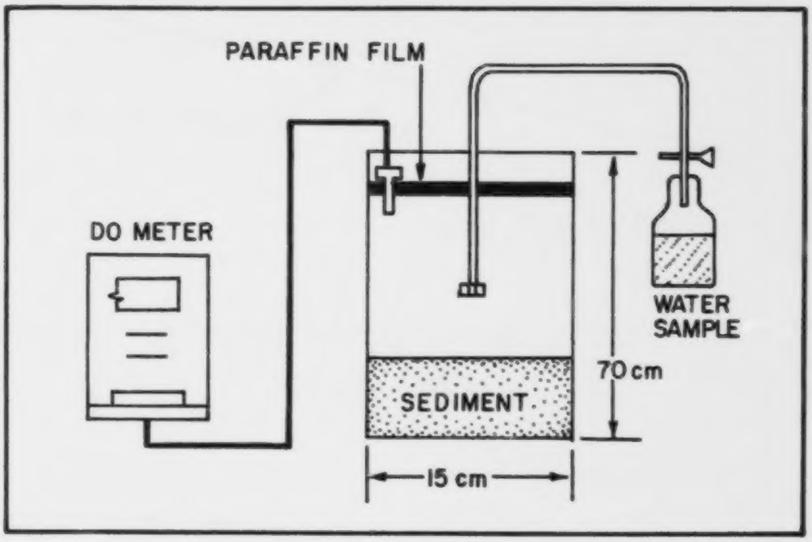


Figure 12. Apparatus for release test procedures.

PARTITION COEFFICIENT

If the relationship between phosphorus release rates and sediment depth is known by measurement, we can derive the values of the partition coefficient ξ, which are calculated back by Equation (16):

$$P_r = \frac{26\ \xi \left\{ P_\beta\ Z + \dfrac{Z_\beta^{\ 3} - (Z_\beta - Z)^3}{6f} \right\}}{(1 + e_o)(1 + \frac{Z}{H})} \qquad \text{as, mg/m}^2\text{/day} \qquad (16)$$

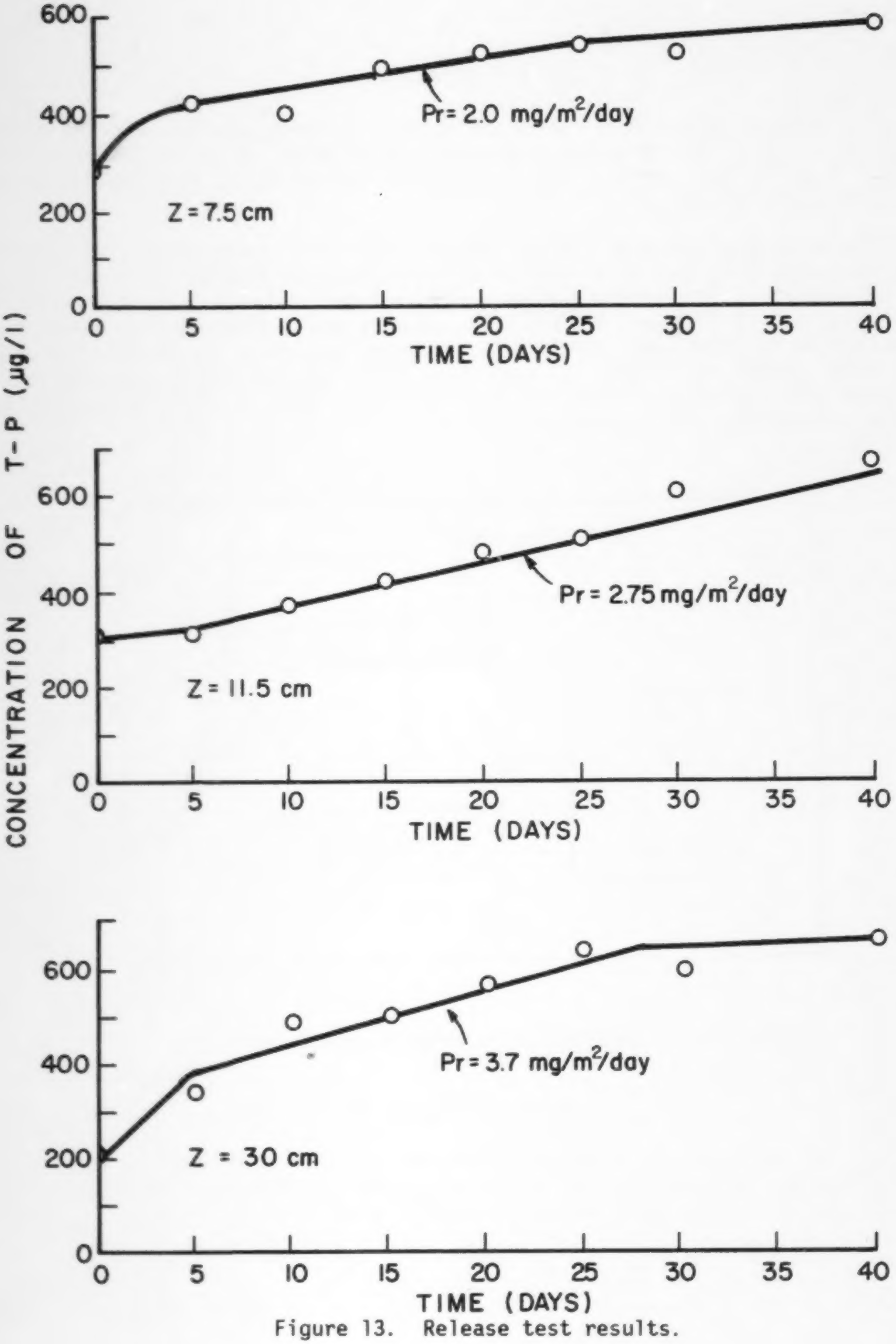


Figure 13. Release test results.

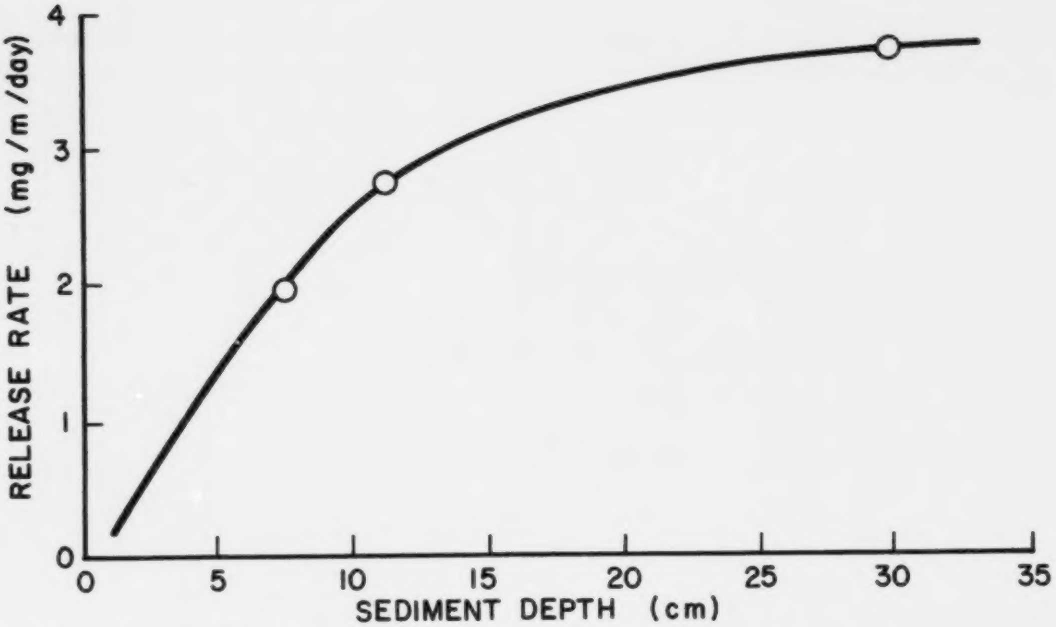


Figure 14. Relationship between phosphorus release rates and sediment depth.

Using the following data:

P_o = 1600 mg/kg

P_β = 500 mg/kg

Z_β = 80 cm

$$f_p = \frac{Z_\beta{}^2}{2\ (P_o - P_\beta)} = \frac{80^2}{2 \times 1100} = 2.909 \quad \text{cm}^2/\text{mg}^{-1}/\text{kg}$$

H = 31.15 cm

Pr_1 = 2.0 mg/m^2/day, for Z = 7.5 cm

Pr_2 = 2.75 mg/m^2/day, for Z = 11.5 cm

Pr_3 = 3.70 mg/m^2/day, for Z = 30.0 cm

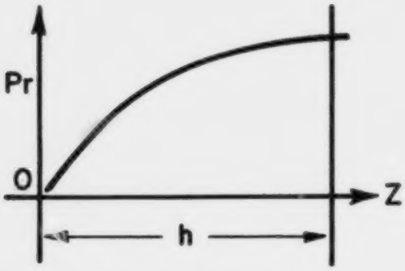
we have:

$\xi_1 = 4.21 \times 10^{-5}$

$\xi_2 = 4.34 \times 10^{-5}$ (day)$^{-1}$

$\xi_3 = 3.76 \times 10^{-5}$

From data shown in Figure 14, it appears that the release rate curve becomes flat at a certain sediment depth.



Thus, the following may be assumed:

$$Z = h \quad ; \quad \frac{\partial Pr}{\partial Z} = 0$$

Differentiating the given equation:

$$P_r = \frac{26\ \xi \left\{ P_\beta Z + \dfrac{Z_\beta{}^3 - (Z_\beta - Z)^3}{6f_p} \right\}}{(1 + e_o)(1 + \frac{Z}{H})} \tag{16}$$

we have:

$$\left| \frac{(1+\frac{Z}{H}) \left[\dfrac{\partial \xi}{\partial Z} \left\{ P_\beta Z + \dfrac{Z_\beta{}^3-(Z_\beta-Z)^3}{6f_p} \right\} + \xi \left\{ P_\beta + \dfrac{(Z_\beta-Z)^2}{2f} \right\} - \dfrac{1}{H} \left\{ P_\beta Z + \dfrac{Z_\beta{}^3-(Z_\beta-Z)^3}{6f_p} \right\} \right]}{(1 + \frac{Z}{H})^2} \right|_{Z=h} = 0$$

Then we obtain:

$$\frac{\partial \xi}{\partial Z} + \alpha\ \xi = 0 \tag{17}$$

where:

$$\alpha = \frac{P_\beta - \dfrac{1}{6f_p H} \left\{ Z_\beta{}^3 - (Z_\beta - h)^3 - 3(H + h)(Z_\beta - h)^2 \right\}}{(1 + \frac{h}{H}) \left\{ P_\beta h + \dfrac{Z_\beta{}^3 - (Z_\beta - Z)^3}{6f_p} \right\}} \tag{18}$$

The solution for Equation (16) is

$$\xi = Ce^{-\alpha Z} \tag{19}$$

This demonstrates that the partition coefficient ξ behaves exponentially against sediment depth.

Since it seems from Figure 14, that h lies at about 30 cm, we have the value of α for h = 30 cm. In Figure 15 the theoretical values of ξ are compared to actual measurements. We can see they are in close agreement, with little deviation.

The theoretical values of phosphorus release (Pr), which are calculated by the function $\xi = Ce^{-0.00866Z}$ based on the point Z = 30 cm, becomes as follows

Sediment Depth cm	Calculated $mg/m^2/day$	Measured $mg/m^2/day$
Z = 7.5	Pr = 2.17	Pr = 2.0
Z = 11.5	Pr = 2.79	Pr = 2.75
Z = 30.0	Pr = 3.75	Pr = 3.75

The comparison in terms of release rates gives an impression of better agreement (Figure 16).

As mentioned above, the close agreement of the calculated and measured values of ξ verifies our theory. So we could establish the following two points in regard to ξ:

1) The partition coefficient ξ lies in the order of 10^{-5} 1/day under anoxic conditions.

2) It behaves as an exponential function in terms of sediment depth. $\xi = Ce^{-\alpha Z}$ That is, its value is greater at the surface and smaller at deeper depths. This means that partitioning of phosphorus is more active in the surface than in the deeper layer.

DETERMINATION OF h

If the release rates are measured, the values of ξ can be calculated by the following equation.

$$\xi = \frac{Pr\,(1 + e_o)\,(1 + \frac{Z}{H})}{26\,\left\{P_\beta Z + \frac{Z_\beta^{\,3} - (Z_\beta - Z)^3}{6f}\right\}}$$

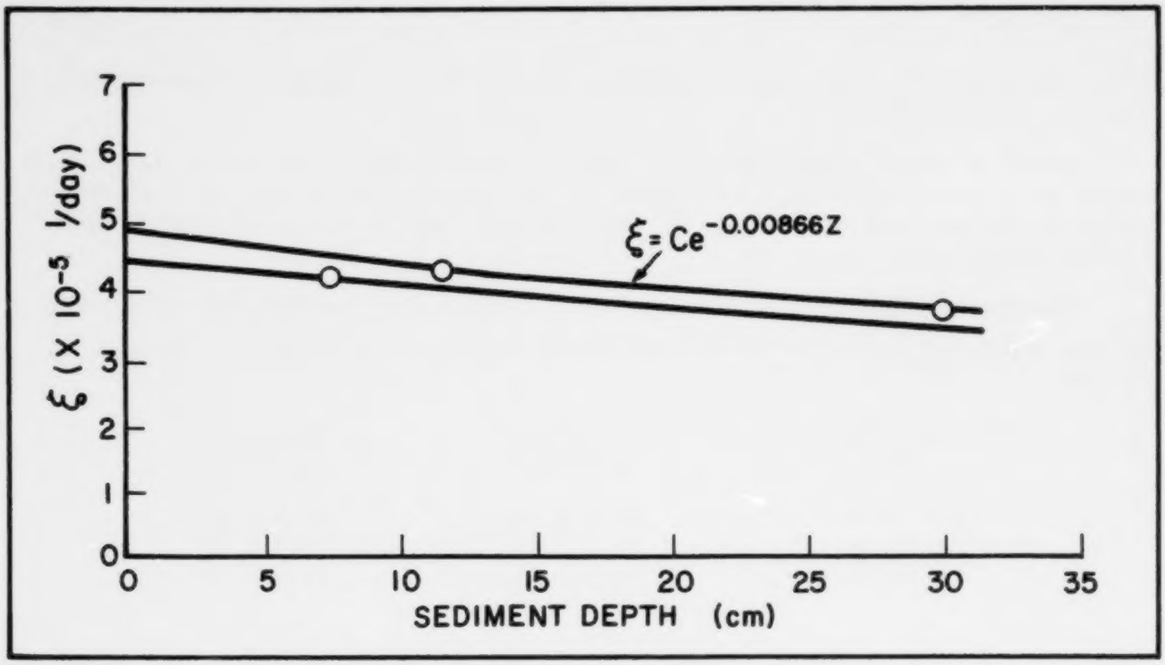


Figure 15. ξ curve.

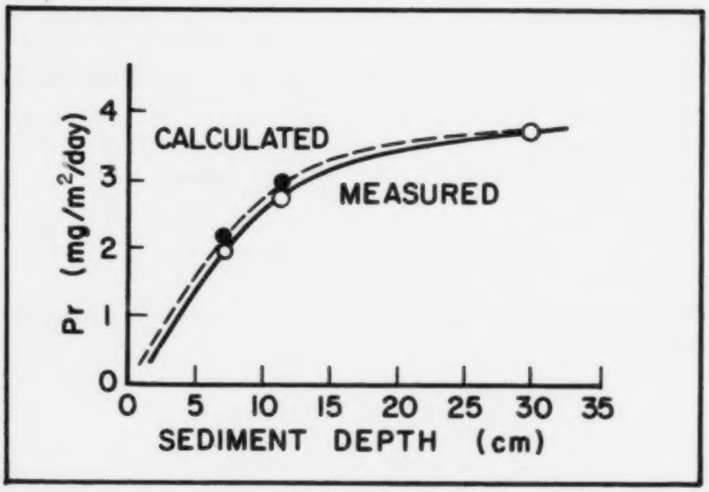


Figure 16. Comparison of calculated and measured phosphorus release.

These are independent of h, but the theoretical values of ξ obtained by $\xi = Ce^{-\alpha Z}$ are dependent on h, because α is a function of h. Therefore, the good agreement of them determines the value of h. As noted previously, it is 30 cm in this case. Figures 17 and 18 indicate that improper assumptions of h will result in significant deviations from the measured values of ξ.

CONCLUSIONS

This paper supports several conclusions regarding phosphorus dynamics:

1) Release rates depend primarily upon the phosphorus content in the sediment.

2) The relationship between release rate and sediment depth is apparently parabolic.

3) Partition coefficients of phosphorus in sediment are on the order of 10^{-5} 1/day in an anoxic condition.

4) Partitioning of phosphorus particulates is more active at the sediment surface than at deeper depths.

5) The sediment depth h, where the release rate curves become flat, is about 20 cm in this case, and perhaps near this in other sediments.

In this study we found that the various indexes and curves obtained showed considerable similarity and order. This implies that we must pay more attention to hydrostatic factors among the various aspects affecting phosphorus release mechanisms.

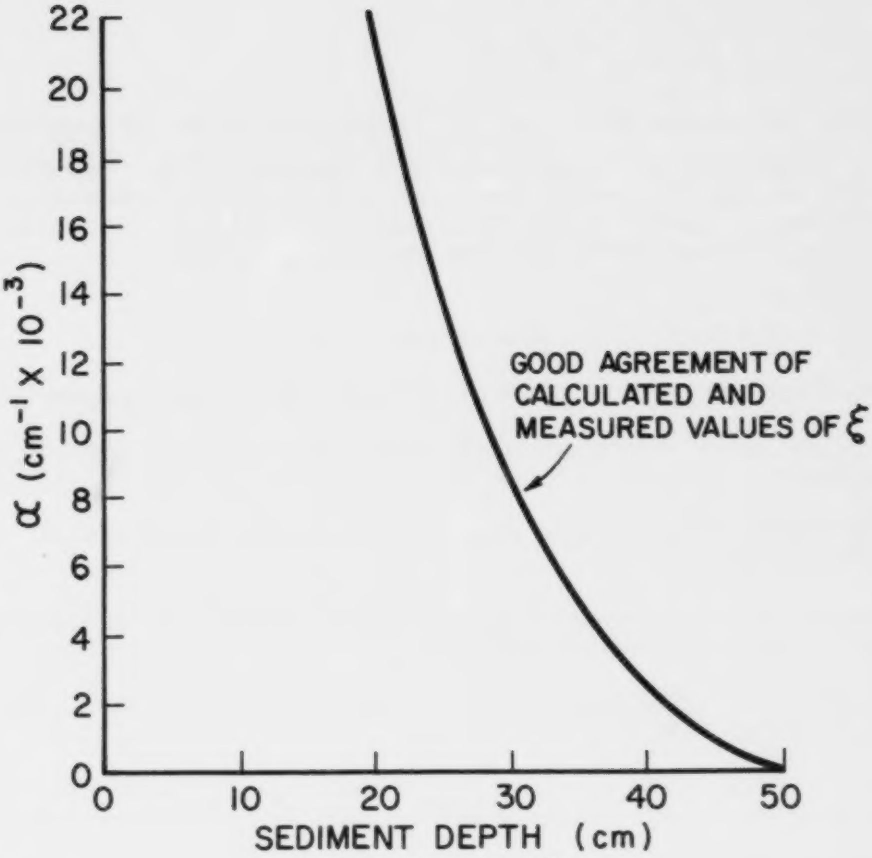


Figure 17. Function of α.

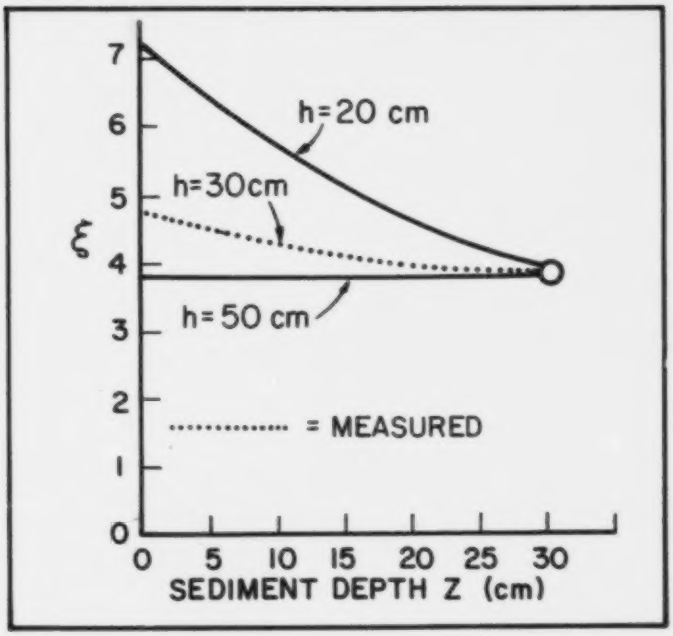


Figure 18. Behavior of ξ against Z.

CONTAINMENT AREA DESIGN FOR SEDIMENTATION OF FINE-GRAINED DREDGED MATERIAL

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ABSTRACT

Procedures are given for containment area design for retention of suspended solids based on solids removal through gravity sedimentation. Separate design procedures for freshwater and saltwater sediments provide for determination of the respective surface area or detention time required to accommodate continuous dredged material disposal. Procedures are also given for estimation of the storage volume required for a single disposal activity and the corresponding ponding depths, freeboard requirements, and dike heights. Laboratory testing procedures required to obtain data for sediment characterization, containment area design, and estimates of long term storage capacity are given. Sediment characterization tests include salinity determination of near bottom water and natural water content, Atterberg limits, organic content, specific gravity, and grain size analysis of the sediments. Sedimentation tests performed in an 8 inch diameter column are used to define settling behavior. Procedures for both flocculent settling tests, generally applicable to freshwater sediments, and zone settling tests, generally applicable to saltwater sediments, are described.

INTRODUCTION

Confinement of dredged material on land has been a disposal alternative used by the Corps of Engineers for a number of years. In more recent years this practice has increased, and added requirements have been placed on the solids retention capability of confined disposal areas. The confined disposal (containment) areas used for both retention and disposal of dredged material are simply sedimentation basins.

Dredged material sedimentation basins are slightly different from those used in water and wastewater treatment in that the dredged material basins must provide for sedimentation to achieve acceptable effluent quality while providing storage volume for several years of material dredged from local waterways. In most cases, the amount of dredged material storage required is probably the controlling factor in sizing a conventional disposal area. Nevertheless, effluents from the large areas now in existence often have problems meeting the effluent requirements for suspended solids. This shortcoming can be attributed to the non-uniform lateral distribution of flows and short-circuiting currents that occur in most dredged material containment areas. As a result of short-circuiting currents, one section of flow is subjected to a different velocity from another. Since sufficient detention time is not provided in this section, the effluent has a higher solids level. This can be seen in aerial photographs of containment areas in operation. An example of a poorly designed containment area is shown in Figure 1. The flow in this containment area is essentially overland flow resulting in significant short-circuiting. In addition, the effective settling area appears to be reduced to above one half the diked area because of a build-up of the coarse-grained dredged material. These factors result in poor suspended solids removal as indicated by the turbidity plume from the weir.

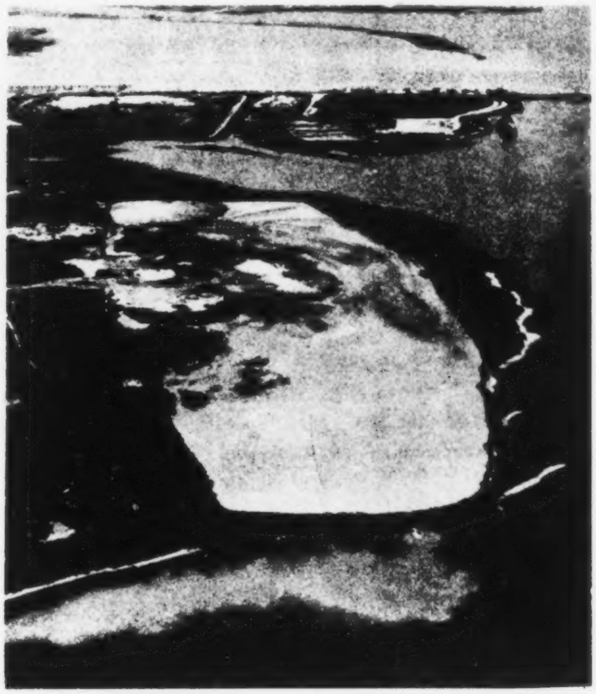


Figure 1. Turbidity from dredged material containment area.

The major problem is that very little is known about the actual sedimentation process in dredged material containment areas. The hydrodynamic problem of one particle falling through a fluid has been solved (Stoke's Law), and formulas have been developed by researchers to determine the fall speed when the density of a particle is very small and their distance apart is much greater than their size. In practice, dredged material is discharged into the sedimentation basin at concentrations averaging about 145 g/l. Because of this high concentration, it is believed that sedimentation occurs under either flocculent or zone settling processes.

High density slurries have been observed near the surface of dredged material sedimentation basins indicating that hindered settling occurs in a significant portion of the water column. The velocity for hindered settling is less than that predicted by theories based on discrete settling because of the upward velocity of water displaced by the highly concentrated slurry. A review of present practices indicates that many dredging disposal operations cannot be undertaken on a continuous basin and still maintain acceptable suspended solids removal levels. Where strict effluent suspended solids limits are enforced, periods of interrupted dredging are common to reduce the loading rate and provide time for particle settling. These interrupted dredging operations usually result in increased overall operational costs.

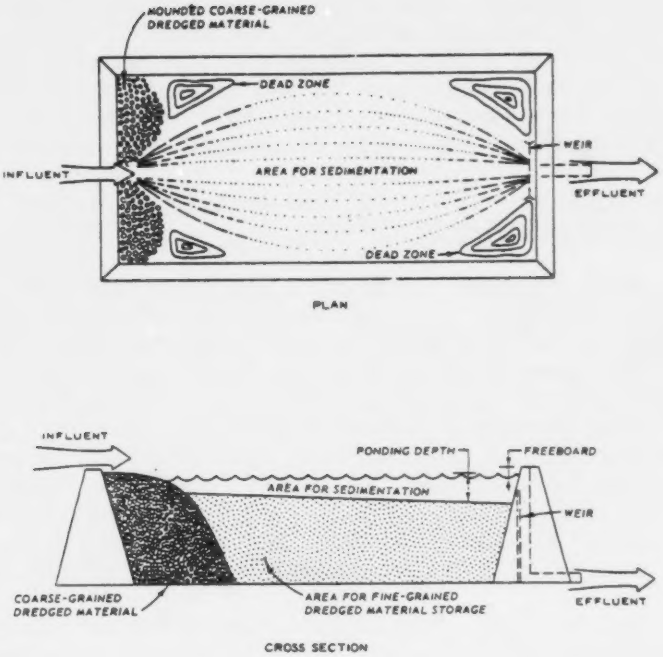
This paper provides procedures for designing fine grained dredged material sedimentation basins to provide adequate retention of suspended solids so that required effluent suspended solids levels can be met. The procedures described herein were developed by the author with funds from the Dredged Material Research Program, Environmental Laboratory, U.S. Army Engineers Waterways Experiment Station, Vicksburg, Mississippi.

CONCEPTS OF CONTAINMENT AREA OPERATION

Diked containment areas are used to retain dredged material solids while allowing the carrier water to be released from the containment area. The two objectives inherent in the design and operation of a containment area are: (a) to provide adequate storage capacity to meet dredging requirements; and (b) to attain the highest possible efficiency in retaining solids during the dredging operation in order to meet effluent suspended solids requirements. These conditions are basically interrelated and depend upon effective design, operation, and management of the containment area.

The major components of a dredged material containment area are shown schematically in Figure 2. A tract of land is surrounded by dikes to form a confined surface area, and the dredged channel sediments are then pumped into this area hydraulically. The influent dredged material slurry can be characterized by suspended solids concentration, particle gradation, type of carrier water (fresh or saline), and rate of inflow.

In some dredging operations, especially in the case of new work dredging, sand, clay balls, and/or gravel may be present. This coarse material (more than half > No. 200 sieve) rapidly falls out of suspension near the dredge inlet pipe forming a mound. The fine grained material (more than half < No.



PLAN

CROSS SECTION

Figure 2. Example of dredged material containment area.

200 sieve) continues to flow through the containment area with most of the solids settling out of suspension, thereby occupying a given storage volume. The fine grained dredged material is usually rather homogeneous and is easily characterized.

The clarified water is discharged from the containment area over a weir. This effluent may be characterized by its suspended solids concentration and rate of outflow. Effluent flow rate is approximately equal to influent flow rate for continuously operating disposal areas. Flow over the weir is controlled by the static head and the effective weir length provided. To promote effective sedimentation, ponded water is maintained in the area; the depth of water is controlled by the elevation of the weir crest. The thickness of the dredged material layer increases with time until the dredging operation is completed. Minimum freeboard requirements and mounding of coarse grained material result in a ponded surface area smaller than the total surface area enclosed by the dikes. Dead spots in corners and other hydraulically inactive zones reduce the effective surface area, where sedimentation takes place, to considerably less than the ponded surface area (1).

FIELD INVESTIGATIONS

Field investigations are necessary to provide data for containment area design. The channel must be surveyed to determine the volume of material to be dredged, and channel sediments must be sampled to obtain material for laboratory tests. This part of the paper describes field investigations required to obtain the necessary samples for laboratory testing. The methods in common use for determining volumes of channel sediment to be dredged are well known and do not warrant discussion here.

The level of effort required for channel sediment sampling is highly project dependent. In the case of routine maintenance work, data from prior samplings and dredging activities can provide a basis for developing the scope of field investigations. Grab samples are considered adequate for sampling fine grained sediments from maintenance dredging locations (2). Such samples are adequate for sediment characterization purposes and are relatively easy and inexpensive to obtain. An evaluation of dredged material after being removed by the hydraulic dredge indicated that grab samples were adequate to characterize the sediment properties (3).

Research by Bartos (4) summarized equipment available for sampling channel sediments. He concluded that there are two general classes of sampling equipment available for use in sampling channel sediments--grab samplers and tube samplers.

Petersen Dredge

The Petersen dredge was found to be adequate for most sampling needs. An example of this type of grab sampler being used is shown in Figure 3. The Petersen dredge is a versatile sampler; it will sample a wide range of bottom texture, from fine grained clays to sands. The Petersen dredge samples 144 square inches to a depth of about 12 inches, depending on the texture of the sediment. It can be seen in Figure 3 that the sampler closes tightly, minimizing the loss of sediment and water upon retrieval. The fine grained sediment samples obtained with this type grab sampler are considered to be representative of *in situ* moisture contents.

Water Samples

Water samples should be taken at the same time as channel sediment samples. The water samples should be taken near the water-sediment interface and used to determine the salinity of the sediment environment. As will be discussed later, salinity levels play an important role in the way sediments settle.

Quantity of Sediment Samples

The quantity of sediment required is based on the amount needed for the laboratory tests. Enough sediment to perform the necessary characterization tests and provide material for the column settling tests should be collected from each established sampling station. Five gallon containers can be used to hold the sediment samples. These containers are about the largest that can be

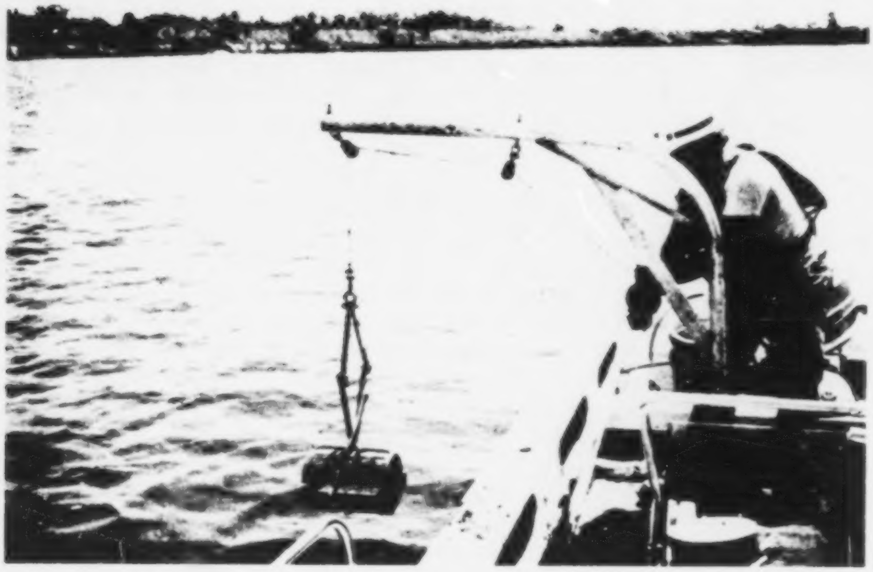


Figure 3. Petersen dredge being used to sample channel sediments.

handled efficiently. Small samples of sediment should be collected and placed in 8-ounce watertight jars for water content and specific gravity tests. Care must be taken to collect the small sediment samples that appear to be most representative of the entire sample.

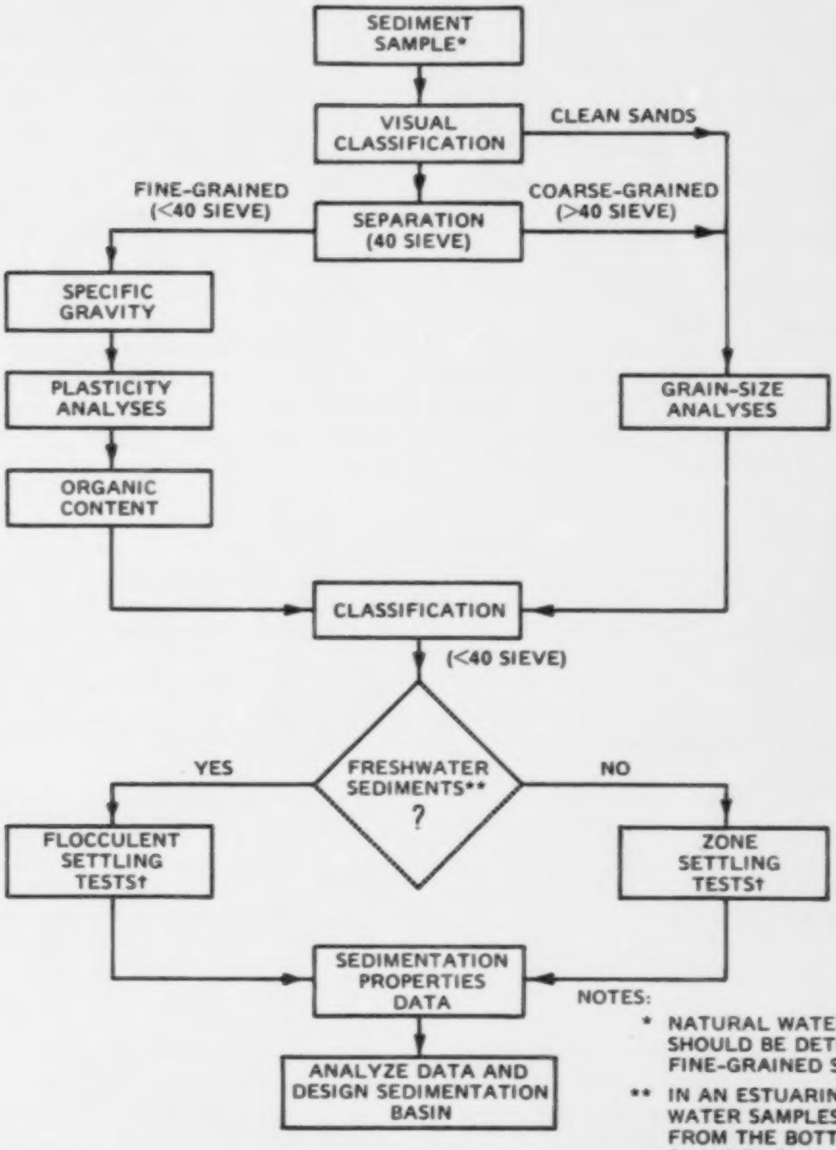
After the characterization tests are performed on grab samples from each sampling station, the samples collected in the 5-gallon containers can be combined to obtain sufficient material for the column settling tests.

Sample Preservation

Samples should be placed in air- and watertight containers and then in a cold room (6 to 8°C) within 24 hours after sampling. The organic content should be determined for each sample and, if less than about 10 percent, it is not considered necessary to have the samples remain in the cold room. Below this organic content level, it is assumed that little biological activity could occur that would affect subsequent testing.

LABORATORY TESTING

Laboratory tests are required primarily to provide data for sediment characterization and containment area design. A flow chart illustrating the complete laboratory testing program for sediment samples is shown in Figure 4.



NOTES:

* NATURAL WATER CONTENTS SHOULD BE DETERMINED ON FINE-GRAINED SEDIMENTS.

** IN AN ESTUARINE SYSTEM WATER SAMPLES TAKEN FROM THE BOTTOM OF THE CHANNEL SHOULD BE TESTED TO DETERMINE SALINITY.

† MAY BE PERFORMED ON COMPOSITE SAMPLES.

Figure 4. Flow chart of recommended laboratory testing program.

Sediment character and requirements for sedimentation data estimates will dictate which laboratory tests are required. Not all laboratory tests indicated in Figure 4 are required for every application. The required magnitude of the laboratory testing program is highly project dependent. Fewer tests are usually required when dealing with a relatively homogeneous material and/or when data are available from previous tests and experience, as is frequently the case in maintenance work. For unusual maintenance projects where considerable variation in sediment properties is apparent from samples or for new work projects, more extensive laboratory testing programs are required. Laboratory tests should always be performed on representative samples selected using sound engineering judgment.

Sediment Characterization Tests

A number of sediment characterization tests are required before settling tests can be performed. Visual classification will establish whether the sediment sample is predominantly fine grained. Tests required on fine grained sediments include natural water content, Atterberg limits, organic content, and specific gravity. The coarse grained sediments require only grain size analyses. Water samples taken from the channel should be tested for salinity to provide information for use in performing the column settling tests.

Sedimentation Tests

Sedimentation, as applied to dredged material disposal activities, refers to those operations in which the dredged material slurry is separated into more clarified water and a more concentrated slurry. Laboratory sedimentation tests must provide data for designing the containment area to meet effluent suspended solids criteria and to provide adequate storage capacity for the dredged solids. These tests are based on the gravity separation of solid particles from the transporting water.

The sedimentation process can be categorized according to three basic classifications: (a) discrete settling where the particle maintains its individuality and does not change in size, shape, or density during the settling process; (b) flocculent settling where particles agglomerate during the settling period with a change in physical properties and settling rate; (c) zone settling where the flocculent suspension forms a lattice structure and settles as a mass, exhibiting a distinct interface during the settling process.

The important factors governing the sedimentation of dredged material solids are initial concentration of the slurry and flocculating properties of the solid particles. Because of the high influent solids concentration and the tendency of dredged material fine grained particles to flocculate, either flocculent or zone settling behavior governs sedimentation in containment areas (3). Discrete settling describes the sedimentation of sand particles and fine grained sediments at concentrations much lower than those found in dredged material containment areas.

Laboratory tests are necessary to characterize the sediment and to provide data for containment area design. A flow chart of the laboratory testing program recommended for providing design data is shown in Figure 4. The recommended laboratory procedures discussed here are for characterization of the dredged material sedimentation process. They are based on results from the extensive laboratory testing program (3, 5).

The objective of running settling tests on sediments to be dredged is to define, on a batch basis, settling behavior in a large scale, continuous flow dredged material containment area. Results of tests must allow determination of numerical values for the design parameters which can be projected to the size and design of the containment area.

Sedimentation of freshwater sediments at slurry concentrations as high as 175 g/l can be characterized by flocculent settling properties (3). However, as slurry concentrations are increased, the sedimentation process may be characterized by zone settling properties. The settling column shown in Figure 5 can be used with procedural modifications for both flocculent and

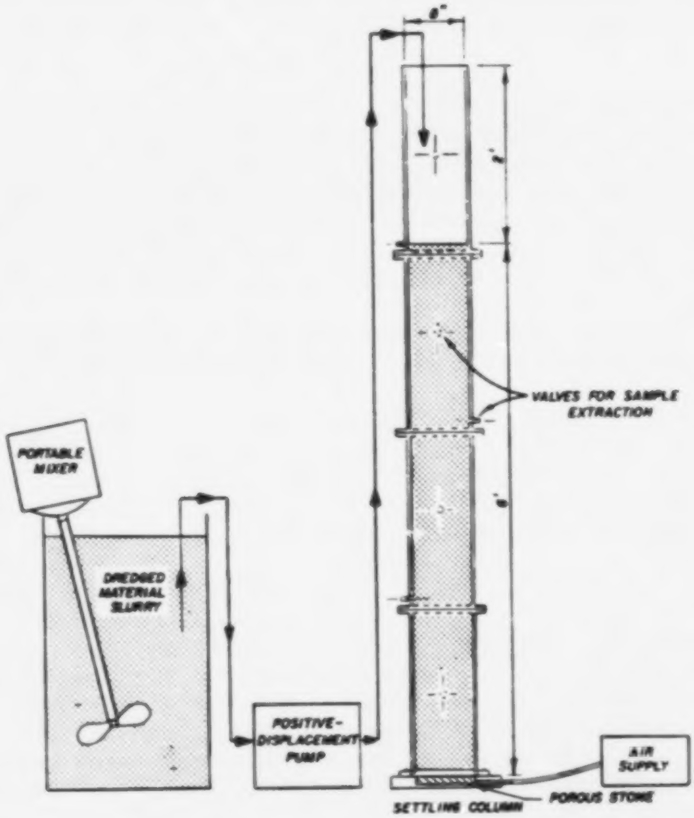


Figure 5. Schematic of flocculent settling test equipment.

zone settling tests. Salinity enhances the agglomeration of dredged material particles (6). The settling properties of all saltwater dredged material tested during the study by Montgomery (5) would be characterized by zone settling tests.

Flocculent Settling Test

The flocculent settling test consists of measuring the concentration of suspended solids at various depths and time intervals in a settling column. If an interface forms near the top of the settling column during the first day of the test, sedimentation is governed by zone settling and that test procedure should be initiated. Information required to design a containment area in which flocculent settling governs can be obtained using a procedure described below.

1. A settling column such as that shown in Figure 5 is used. The test column depth should approximate the effective settling depth of the proposed containment area. A practical test depth is 6 feet. The column should be at least 8 inches in diameter with sample ports at 1-foot intervals. The column should have provisions to bubble air from the bottom to keep the slurry mixed during the column-filling period.

2. Mix the sediment slurry to the desired suspended solids concentration in a container with sufficient volume to fill the test column. At least two tests should be performed at the concentration selected to represent the concentration of influent dredged material C_i. Use the average detention time computed from these tests for design. Field studies indicate that for maintenance dredging in fine grained material the disposal concentrations average about 145 g/l.

3. Pump or pour the slurry into the test column using air to maintain a uniform concentration during the filling period.

4. While the column is completely mixed, draw off samples at each sample port and determine the suspended solids concentration. Average these values and use the results as the initial concentration at the start of the test. After the initial samples are taken, stop the air bubbling and begin the test.

5. Allow the slurry to settle, then withdraw samples from each sampling port at regular time intervals and determine suspended solids concentrations. Sampling intervals depend on the settling rate of the solids--usually at 30-minute intervals for the first 3 hours and then at 4-hour intervals until the end of the test. The sampling times can be adjusted after the first complete test. Continue the test until the interface of solids can be seen near the bottom of the column and the suspended solids level in the fluid above the interface is < 1 g/l.

6. If an interface has not formed within the first day on any previous tests, run one additional test with a suspended solids concentration sufficiently high to induce zone settling behavior. This test should be carried out according to the procedures outlined below for zone settling tests. The exact concentration at which zone settling behavior occurs depends upon the sediment being used to estimate the volume required for dredged material storage.

Zone Settling Test

The zone settling test consists of placing a slurry in a sedimentation column and recording the fall of the liquid-solids interface with time. Plot the depth to the interface versus time as illustrated in Figure 6. The slope of the constant settling zone of the curve is the zone settling velocity, which is a function of the initial test slurry concentration. Information required to design a containment area in which zone settling governs can be obtained by using the procedure described below.

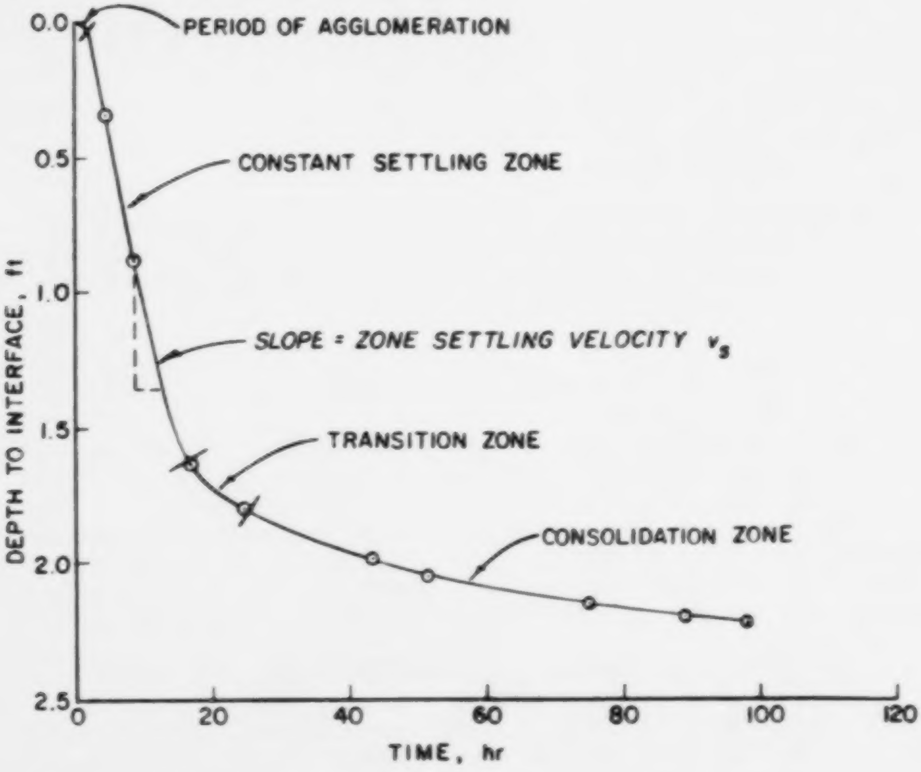


Figure 6. Typical batch settling curve for dredged material.

1. Use a settling column such as that shown in Figure 5. It is important that the column diameter be sufficient to reduce wall effects and the test be performed at a slurry depth near that expected in the field. Therefore, a one-liter graduated cylinder should never be used to perform a zone settling test for sediment slurries representing dredged-disposal activities.

2. Mix the slurry to the desired concentration and pump or pour it into the test column. Test concentrations should range from about 60 to 200 g/l. Air may not be necessary to keep the slurry mixed if the filling time is less than 1 minute.

3. Record the depth to the solid-liquid interface with respect to time. Observations must be made at regular intervals to gain data for plotting the depth to interface versus time curve. It is important to make enough observations to clearly define this curve for each test.

4. Continue the readings until sufficient data are available to define the maximum point of curvature of the depth to interface versus time curve for each test. The tests may require from 1 to 5 days to complete.

5. Perform a minimum of eight tests. Data from these tests are required to develop the curve of zone settling velocity versus concentration.

6. One of the above tests should be performed on sediment slurries at a concentration of about 145 g/l. The test should be continued for a period of at least 15 days to provide data for estimating volume requirements.

DESIGN PROCEDURES

The flow chart shown in Figure 7 illustrates the design procedures recommended in the following paragraphs. The design procedures were adapted from procedures used in water and wastewater treatment and are based on field and laboratory investigations on sediments and dredged material (3). Design methods for saltwater and freshwater sediments are presented. Essentially the method for saltwater sediments is based on zone settling properties, and the method for freshwater sediments is based on flocculent settling properties.

The design procedures presented here are for gravity sedimentation of dredged suspended solids. However, gravity sedimentation will not completely remove suspended solids from containment area effluent since wind and other factors can resuspend solids and increase effluent solids concentration. The sedimentation process, with proper design and operation, will normally provide removal of fine grained sediments down to levels of 1 and 2 g/l in the effluent for saltwater and freshwater sediments, respectively. If the required effluent standards are lower than this, the designer must provide for additional treatment of the effluent, e.g., flocculation or filtration.

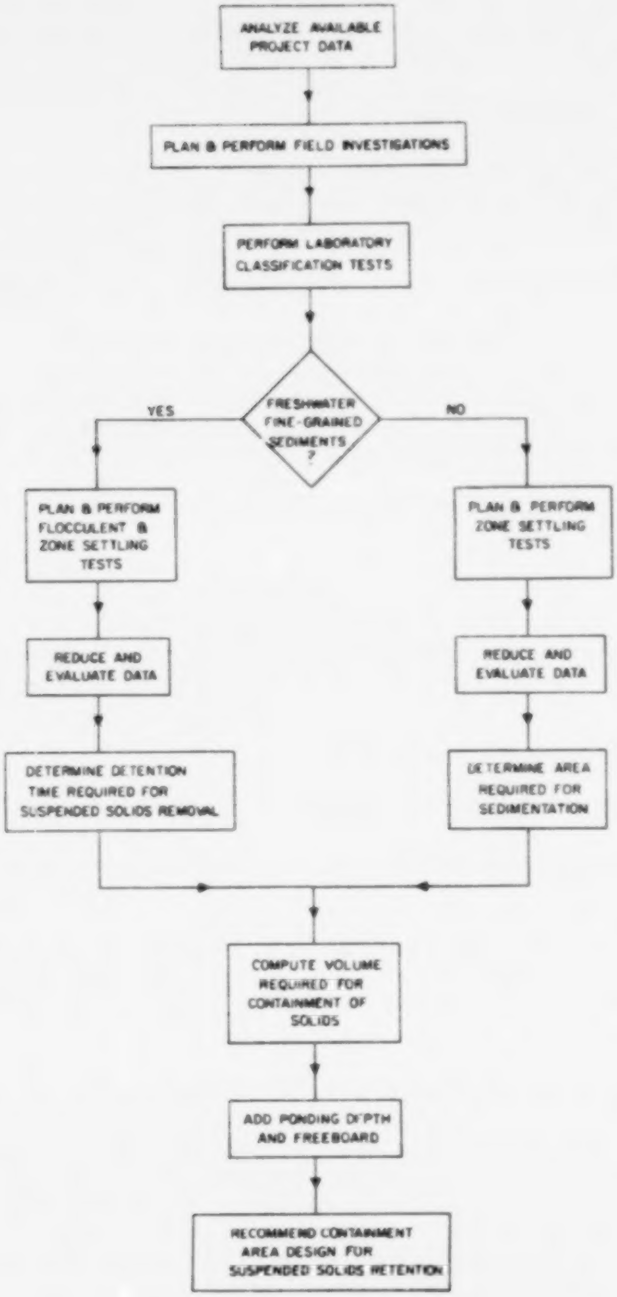

ANALYZE AVAILABLE PROJECT DATA
PLAN & PERFORM FIELD INVESTIGATIONS
PERFORM LABORATORY CLASSIFICATION TESTS
FRESHWATER FINE-GRAINED SEDIMENTS ?
YES
NO
PLAN & PERFORM FLOCCULENT & ZONE SETTLING TESTS
PLAN & PERFORM ZONE SETTLING TESTS
REDUCE AND EVALUATE DATA
REDUCE AND EVALUATE DATA
DETERMINE DETENTION TIME REQUIRED FOR SUSPENDED SOLIDS REMOVAL
DETERMINE AREA REQUIRED FOR SEDIMENTATION
COMPUTE VOLUME REQUIRED FOR CONTAINMENT OF SOLIDS
ADD PONDING DEPTH AND FREEBOARD
RECOMMEND CONTAINMENT AREA DESIGN FOR SUSPENDED SOLIDS RETENTION

Figure 7. Flow chart of recommended design procedures for fine-grained sediments.

Data Requirements

The data required to use the design procedures are obtained from field investigations, laboratory testing, dredging equipment designs, and past experience in dredging and disposal activities.

Estimate *In Situ* Sediment Volume

The initial step in any dredging activity is to estimate the *in situ* volume of sediment to be dredged. Sediment quantities are usually determined from channel surveys on a routine basis.

Determine Physical Characteristics of Sediments

Field sampling and laboratory testing should be accomplished according to the methods discussed by Palermo *et al*. (2). Adequate sample coverage is required to provide representative samples of the sediment. *In situ* water contents of the fine-grained sediments are also required. Care must be taken in sampling to ensure that the water contents are representative of the *in situ* conditions. The water content of representative samples, w, is used to determine the *in situ* void ratio e_i as follows:

$$e_i = \frac{(w/100)G_s}{S_d/100} \tag{1}$$

where

w = water content in percent

G_s = specific gravity of sediment solids

S_d = degree of saturation (equal to 100 percent for sediment)

A representative value from *in situ* void ratios is used later to estimate volume for the containment area. Grain-size analyses must be performed to estimate the quantities of coarse and fine-grained material in the sediment to be dredged.

Obtain and Analyze Proposed Dredging and Disposal Data

The designer must obtain and analyze data concerning the dredged material disposal rate. For hydraulic pipeline dredges, the type and size of dredge(s) to be used and the average solids concentration of the dredged material when discharged into the containment area must be considered. If the size of the dredge to be used is not known, the design must assume the largest dredge size that might be expected to perform the dredging.

Based on these data, the designer must estimate or determine containment area influent rate, influent suspended solids concentration, effluent rate (for weir sizing), effluent concentration allowed, and time required to complete the disposal activity. If no other data are available for hydraulic pipeline dredges, an influent suspended solids concentration of 145 g/l (13 percent by weight) can be used for design purposes.

Perform Laboratory Sedimentation Tests

The procedures for sedimentation tests are given earlier in the section on laboratory testing. A designer must evaluate the results of salinity tests to determine whether the sediments to be dredged are freshwater or saltwater sediments. If the salinity is about 3 ppt, the sediments are classified as saltwater sediments for the purpose of selecting the laboratory sedimentation test.

Design Method for Saltwater Sediments

The following design method is recommended for sedimentation of dredged material from a saltwater environment. It can also be used for freshwater dredged material if the laboratory settling tests indicate zone settling properties. An example of this design method is presented by Montgomery (3).

Analyze Laboratory Data

A series of zone settling tests must be conducted as detailed earlier. The results of the settling tests are analyzed to determine zone settling velocities at the various suspended solids concentrations. The procedure is as follows:

1. Develop a settling curve for each test. Plot depth to interface versus time.

2. Calculate the zone settling velocity, v_s, as the slope of the constant settling zone (straight line portion of the curve). The velocity should be in feet per hour.

3. Plot v_s versus suspended solids concentrations on a semi-log plot.

4. Use the plot developed in step 3 to develop a solids loading versus solids concentration curve as shown in Figure 8.

Compute Design Concentration

The design concentration, C_D, is defined as the average concentration of the dredged material in the containment area at the end of the disposal activity and is estimated from data obtained from the 15-day column settling tests. The following steps can be used to estimate average containment area concentrations for each 15-day column settling test. It may be desirable to perform more than one 15-day test. If so, use an average of the values as the design concentration.

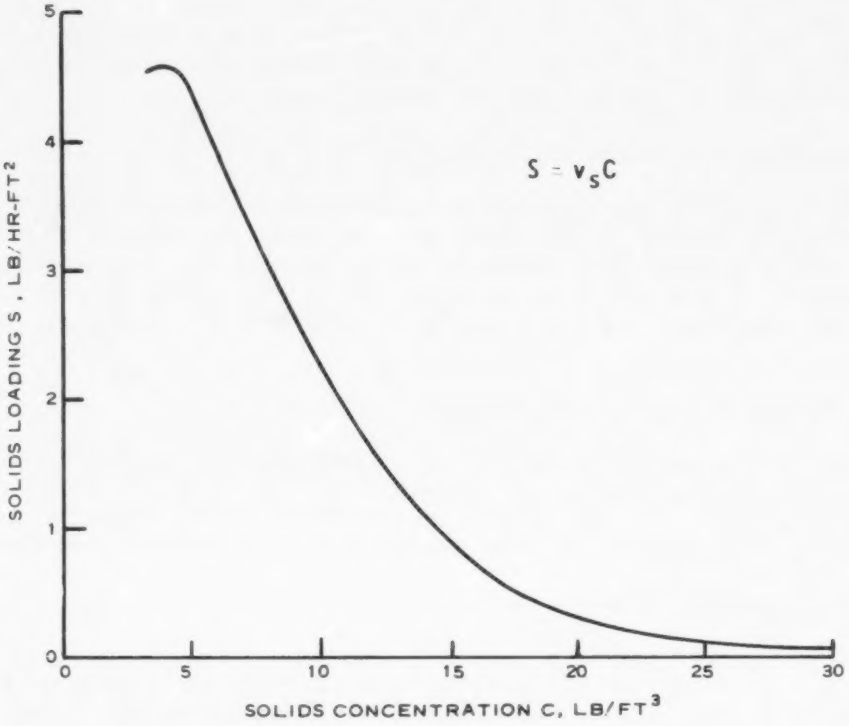


Figure 8. Typical solids loading curve for dredged material.

1. Compute concentration versus time for the 15-day settling test. Assume zero solids in the water above the solids interface to simplify calculation.

2. Plot concentrations versus time on log-log paper (see Figure 9).

3. Draw a straight line through the data points. This line should be drawn through the points representing the consolidation zone.

4. Estimate the time of dredging by dividing the dredge production rate into volume of sediment to be dredged.

5. Estimate the concentration at $t_{1/2}$ (one-half the time required for the disposal activity determined in step 4) using the figure developed in steps 2 and 3. This time is an approximation of the average time of residence for the dredged material in the containment area. Since concentration is a function of time, one-half the dredging

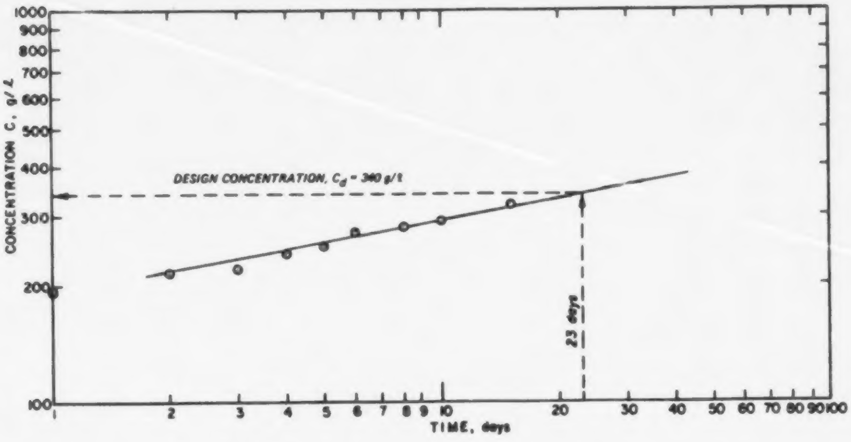


Figure 9. Concentration versus time.

time would represent a period during which one-half of the dredged material would have been in the area longer and the other half less than a time equal to one-half the dredging time.

6. Use the value computed in step 5 as the design solids concentration, C_D.

Compute Area Required for Sedimentation

Containment areas designed according to the following steps should provide removal of fine grained sediments well enough so that suspended solids levels in the effluent do not exceed 1 to 2 g/l. The area required for the zone settling process to concentrate the dredged material to the design concentration is computed as follows, using the Yoshioka et al. (7) graphic solution to the Coe and Clevenger procedure (8).

1. Use the design concentration and construct an operating line from the design solids concentration tangent to the loading curve as shown in Figure 10. The design loading is obtained on the y-axis as S_L.

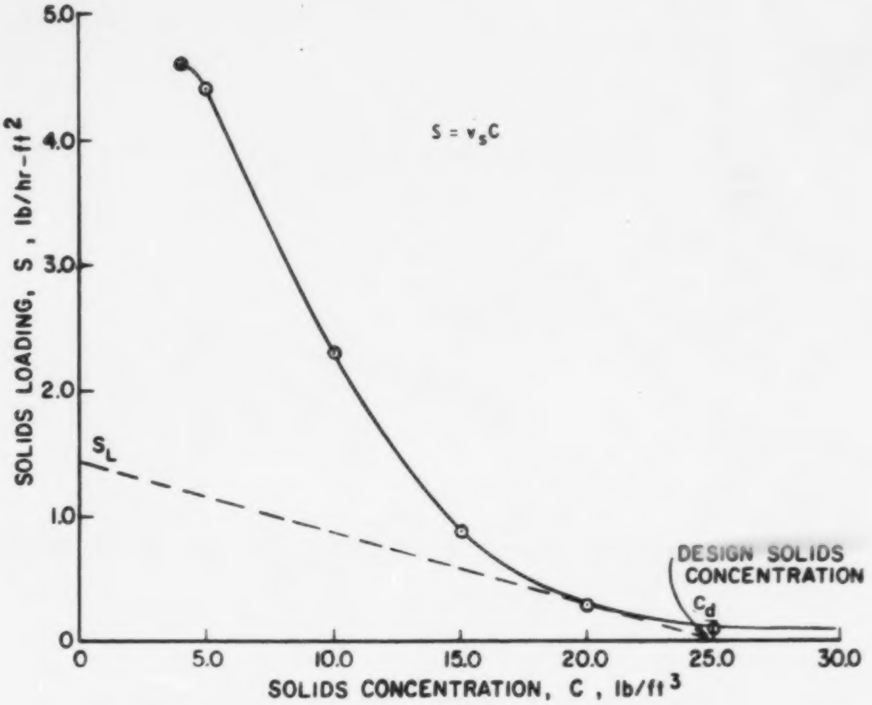


Figure 10. Solids loading curve showing design line.

2. Compute the required area as

$$A = \frac{Q_i C_i}{S_L} \tag{2}$$

where

A = containment area surface requirement, ft^2

Q_i = influent rate, ft^3/hr ($Q_i = A_p V_d$; assume V_d = 15 ft/sec in absence of data and convert Q_i calculated in ft^3/sec to ft^3/hr)

A_p = cross sectional area of dredge pipeline, ft^2

V_d = velocity of dredge discharge, ft/sec

C_i = influent solids concentration, lb/ft^3 (use 145 g/l or 9.2 lb/ft^3 if no data are available)

S_L = design solids loading, $lb/hr-ft^2$

3. Increase area by a factor of 2.25 to compensate for containment area short-circuiting and dispersion.

$$A_d = 2.25\ A \tag{3}$$

where

A_d = design basin surface area, ft^2

A = area determined from Equation 2, ft^2

Design Method for Freshwater Sediments

Sediments in a dredged material sedimentation basin are comprised of a broad range of particle flocs of different sizes and surface characteristics. In the sedimentation basin under flocculent settling conditions the large particle flocs settle at faster rates, thus overtaking finer flocs in their descent. This contact increases the floc sizes and enhances settling rates. The greater the ponding depth in the containment area, the greater is the opportunity for contact among sediments and flocs. Therefore, sedimentation of freshwater dredged sediments is dependent on the ponding depth as well as the properties of the particles.

The results of one flocculent test are shown in Figure 11. The initial concentration of the test slurry was 175 g/l and the depth of slurry was 8 feet. Percent by dry weight of initial concentration was plotted versus depth for various times. These times represent the period of settling for the slurry. Settling data plotted in this manner can be used to evaluate the dominant sedimentation process (9). The dashed lines in Figure 11 were generated by dividing the depth by time and plotting dashed lines of constant d/t values. It is possible to tell directly from this plot that flocculation is causing the particles to settle more rapidly (9). This is indicated by the fact that the dashed lines, d/t, slope toward the d-axis. When neither zone settling nor flocculation occurs, the dashed lines will be straight and parallel to the d-axis. When zone settling slows the particles down more than flocculation can speed them up, the dashed lines will slope away from the d-axis.

Analyze Laboratory Data

Evaluation of the sedimentation characteristics of a freshwater sediment slurry is accomplished as discussed earlier. The design steps are as follows [refer to Montgomery (3) for example problem]:

1. Arrange data from laboratory tests illustrated in Table 1 into the form shown in Table 2.

2. Plot these data as shown in Figure 11. The percent by weight of initial concentration for each depth and time is given in Table 2. The solid curved lines represent the concentration depth profile at

TABLE 1. OBSERVED FLOCCULENT SETTLING CONCENTRATIONS WITH DEPTH (in grams per liter)

Time (min)	Depth from top of settling column (feet)						
	1	2	3	4	5	6	7
0	132	132	132	132	132	132	132
30	46	99	115	125	128	135	146
60	25	49	72	96	115	128	186
120	14	20	22	55	78	122	227
180	11	14	16	29	75	119	
240	6.8	10.2	12	18	64	117	
360	3.6	5.8	7.5	10	37	115	
600	2.8	2.9	3.9	4.4	14	114	
720	1.0	1.6	1.9	3.1	4.5	110	
1020	0.9	1.4	1.7	2.4	3.2	106	
1260	0.8	1.1	1.2	1.4	1.7	105	
1500	0.7	0.9	1.0	1.1	1.2	92	
1740	0.6	0.7	0.8	0.9	1.0	90	

Note: Data from actual test on freshwater sediments. Although a 6-foot test depth is recommended, an 8-foot depth was used in this test.

TABLE 2. PERCENT OF INITIAL CONCENTRATION WITH TIME

Time, T (min)	Depth from top of settling column		
	1 ft	2 ft	3 ft
0	100	100	100
30	35	75	87
60	19	37	55
120	11	15	17
180	8	11	12
240	5	8	9
360	3	4	6
600	2.0	2.2	3.0
720	1.0	1.2	1.4

Note: Initial suspended solids concentration = 132 g/l.

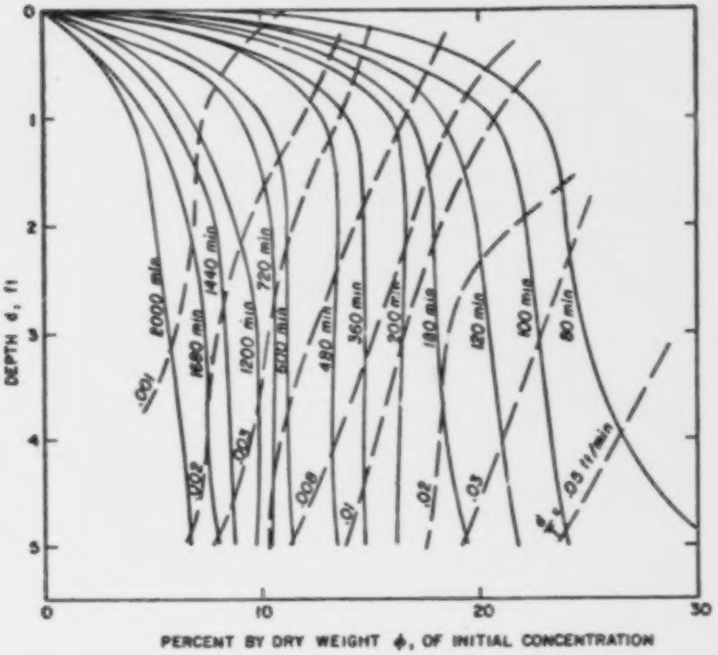


Figure 11. Depth versus percent solids by weight of initial concentration.

various times during settling (refer to Figure 11 for more details). Numbers appearing along the horizontal depth lines are used to indicate area boundaries.

3. Compute a design concentration using data from the 15-day zone settling test. Follow the procedure outlined in the design method for saltwater sediments.

Compute Detention Time Required for Sedimentation

The detention time is computed as follows:

1. Calculate removal percentage at depths of 1, 2, and 3 feet for various times using the plot illustrated in Figure 12. The removal percentage for depth d_1 and t = 1 is computed as follows:

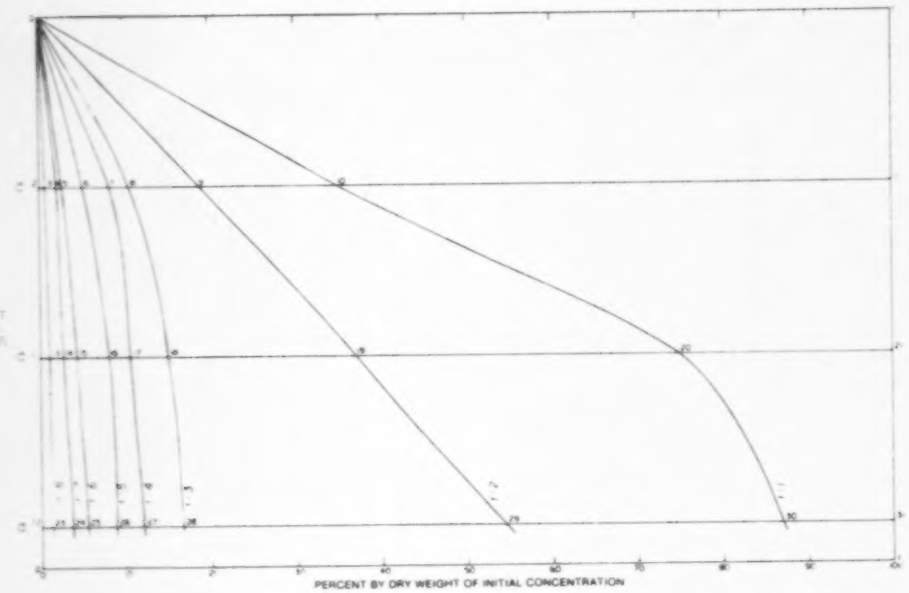


Figure 12. Removal of flocculating dredged material particles.

$$R = \frac{\text{Area } 0,\ 10,\ 11,\ 1^*}{\text{Area } 0,\ 2,\ 11,\ 1} \times 100 \qquad (4)$$

where

R = removal percentage

Determine these areas by either planimetering the plot or by direct graphic measurements and calculations. This appproach is used to calculate removal percentages for each depth as a function of time. The depths used should cover the range of ponding depths expected in the containment area. This report recommends at least 2 feet of ponding depth at the end of the dredging project.

2. Plot the solids removal percentages versus time as shown in Figure 13.

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* These numbers correspond to the numbers used in Figure 12 to indicate the area boundaries for the total area down to depth d (0, 2, 11, 1), and the area to the right of the t = 1 time line (0, 10, 11, 1).

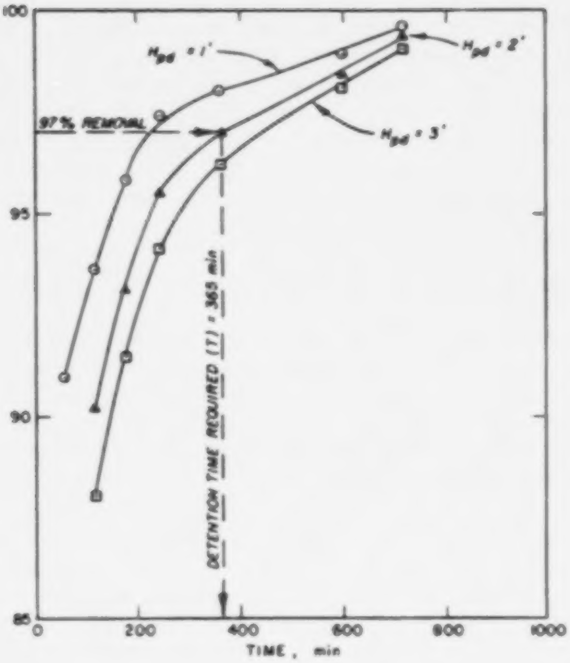

Figure 13. Solids removal versus time as a function of depth.

3. Theoretical detention times can be selected from Figure 13 for various solids removal percentages. Select the detention time, T, that gives the desired removal percentage for the design ponding depth.

4. The theoretical detention time, T, should be increased by a factor of 2.25 to compensate for the fact that sedimentation basins, because of short circuiting and dispersion, have average detention times less than volumetric detention times:

$$T_d = 2.25\ T \qquad (5)$$

where

T_d = design detention time

Volume Requirements for Containment of Solids

The procedures outlined in the above paragraphs are aimed at providing sedimentation basins with sufficient areas and detention times to accommodate continuous disposal activities while providing sufficient suspended solids removal to meet effluent suspended solids requirements. Sedimentation basins must also be designed to meet volume requirements for a particular disposal activity. The total volume required of a sedimentation basin includes volume for storage of dredged material, volume for sedimentation (ponding depths), and freeboard volume (volume above water surface). Volume required for storage of the coarse-grained material (> 200 sieve) must be determined separately, because this behaves independently of the fine-grained (< 200 sieve) material.

Estimate Volume Occupied by Dredged Material in Sedimentation Basin

The volume occupied by dredged material in the sedimentation basins after the completion of a particular disposal activity is computed as follows. The volume is not an estimate of the long-term needs for multiple-disposal activities. The procedures given below can be used to design for volume required for one disposal activity, or used to evaluate the adequacy of the volume provided by an existing sedimentation basin.

1. Compute the average void ratio of fine-grained dredged material in the sedimentation basin at the completion of the dredging operation using the design concentration determined in earlier steps as dry density of solids. (Note that design concentration is determined for both flocculent and the zone sedimentation design procedure.) Use the following equations to determine void ratio:

$$e_o = \frac{G_s \gamma_w}{\gamma_d} - 1 \qquad (6)$$

where

e_o = average void ratio of dredged material in the sedimentation basin at the completion of the dredging operation

G_s = specific gravity

γ_w = density of water, g/l

γ_d = dry density of solids at design concentrations, ($C_D = \gamma_d$) g/l

2. Compute the change in volume of fine-grained channel sediments after disposal in the sedimentation basin from:

$$\Delta V = V_i \frac{e_o - e_i}{1 + e_i} \tag{7}$$

where

ΔV = change in volume of fine-grained channel sediments after disposal in the sedimentation basin, ft^3

e_i = average void ratio of *in situ* channel sediments

V_i = volume of fine-grained channel sediments, ft^3

3. Compute the volume required by dredged material in the sedimentation basin from:

$$V = V_i + \Delta V + V_{sd} \tag{8}$$

where

V = volume of dredged material in the sedimentation basin at the end of the dredging operation, ft^3

V_{sd} = volume of sand (compute using 1:1 ratio), ft^3

Estimate of Basin Depth

Previous calculations have provided a design area A_d and design detention time T_d required for fine-grained dredged material sedimentation. Equations 6-8 are used to estimate the volume and corresponding depth requirements for the storage of solids in the containment area. Throughout the design process, the existing topography of the containment area must be considered, since it can have a significant effect on the average depth of the containment area.

Saltwater Sediments (Zone Settling)

The following procedure should be used for saltwater sediments:

1. Estimate the thickness of dredged material at the end of the disposal operation from:

$$H_{dm} = \frac{V}{A_d} \tag{9}$$

where

H_{dm} = thickness of the dredged material layer at the end of dredging operation, ft

V = volume of dredged material in the basin, ft^3 (from Equation 8)

A_d = design area, ft^2 (as determined from Equation 3 or known surface area for existing sites)

2. Consult with soils design engineers to determine maximum height allowed for confining dikes. Anticipated settlement of the dikes should also be considered.

3. Add ponding depth and freeboard depth to H_{dm} to determine the required containment area depth (dike height).

$$D = H_{dm} + H_{pd} + H_{fb} \tag{10}$$

where

D = dike height, ft

H_{pd} = average ponding depth over the area, ft (a minimum of 2 feet is recommended)

H_{fb} = freeboard above the basin water surface to prevent wave overtopping and subsequent damage to confining earth dikes, ft (a minimum of 2 feet is recommended to account for fetch and wind).

4. Compare with allowable dike height.

Freshwater Sediments (Flocculent Settling)

The following procedure should be used for freshwater sediments:

1. Compute the volume required for sedimentation from:

$$V_B = Q_i T_d \tag{11}$$

where

V_B = sedimentation basin volume required for meeting suspended solids effluent requirements, ft^3

T_d = required detention time from Equation 5

2. Consult with soils design engineers to determine maximum height allowable for confining dikes D. In some cases, it might be desirable to use less than the maximum allowed dike height.

3. Compute the required design area as a minimum required surface area for solids storage from:

$$A_d = \frac{V}{H_{dm(max)}} \tag{12}$$

where

$$H_{dm(max)} = D - H_{pd} - H_{fb}$$

or set the design area A_d equal to the known surface area for existing sites.

4. Evaluate volume available for sedimentation near the end of the disposal operation from:

$$V^* = H_{pd} A_d \tag{13}$$

where

V^* = volume available for sedimentation near the end of disposal operation, ft^3

5. Compare V^* and V_B. If the volume required for sedimentation is larger than V^*, the sedimentation basin will not meet the suspended solids effluent requirements for the entire disposal operation. The following three measures can be considered to ensure that effluent requirements are met: (1) increase the design area, A_d; (2) operate the dredge on an intermittent basis when V^* becomes less than V_B or use smaller size dredge; and (3) provide for post-treatment of effluent to remove solids.

6. Estimate the thickness of dredged material at the end of disposal operation using Equation 9. A_d is determined using step 3 above.

7. Determine the required sedimentation basin depth using Equation 10.

8. Compare with maximum allowable dike height (see paragraph below).

At most sedimentation basins, the foundation soils are soft. Such foundations limit the heights of confining earth dikes that can be economically constructed. Therefore, soils design engineers must be consulted to determine the maximum dike height that can be constructed. If the maximum dike height allowed by foundation conditions is less than the sedimentation basin depth requirement, the design area A_d must be increased until the depth requirement can be accomodated by the allowable dike height; the thickness of the dredged material layer must also be decreased.

SUMMARY

The field verification work performed by Montgomery (5) indicated that conservative values could be estimated from laboratory tests for solids concentrations expected in the dredged material sedimentation basin. The laboratory tests data were reasonably close and should be adequate for design purposes. The column sedimentation tests can be improved with further experience in dredged material sedimentation basin design and with more laboratory testing.

The flocculent settling tests and design procedures recommended for freshwater sediments were found to provide design estimates that agreed closely with actual field values. However, additional cases should be evaluated before full substantiation of these procedures is proclaimed.

A significant amount of work is required on the hydraulic efficiency of dredged material sedimentation basins before firm design correction factors can be established to account for scale-up and flow-through problems. Five dye tracer tests were evaluated by Montgomery (5). The correction factors determined from these tests varied from 2.13 to 2.72. Based on the work accomplished during this research, a correction factor of 2.25 appears reasonable. This factor is higher than those recommended in the sanitary engineering literature. However, the conditions experienced at a dredged material sedimentation basin are more complex than those of wastewater treatment facilities.

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SAMPLING, PRESERVATION, AND ANALYSIS OF SEDIMENT SAMPLES: STATE-OF-THE-ART LIMITATIONS

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ABSTRACT

The collection and analysis of sediment samples has become a topic of increased regulatory concern. This paper presents a procedure for preparing a sediment sampling program based on an awareness of the interrelated nature of the decisions to be made. The first step in the approach is to explicitly state the purpose of the intended study so that testing procedures can be selected based on the sample property they measure. This provides the most efficient use of available procedures and simplifies the selection of sample handling and storage requirements. Selection of sample collection techniques and locations is the most difficult area to provide definitive guidance because of the importance of point sources, local hydrology, study purposes, and financial resources. Therefore, bias that can be introduced into the final data as a consequence of collection technique and sample location decisions are discussed. By being aware of the present limitations of available sampling techniques and analytical procedures, project managers can define the best sampling program for their specific need.

INTRODUCTION

Sediments are an important reservoir in the aquatic cycling of chemical contaminants that can respond to equilibrium stresses. Major point source discharges can enrich the composition of sediments through the processes of precipitation and sorption/settling. The mechanisms of mixing and diffusion can also result in the transfer of sediment-associated constituents from the sediments to the overlying water column. The dynamic, although ill-defined, interaction between sediments and water and the tremendous quantities of material involved in dredging activities has created increased regulatory concern over the composition of sediments and the effects of sediments on water quality. This concern has contributed to the implementation of Section

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404 of Public Law 92-500 (Clean Water Act) and Section 103 of Public Law 92-532 (Marine Protection, Research, and Sanctuary Act). These regulations are requiring a more complete characterization of sediments to be dredged and an evaluation of the potential environmental consequences of disposing of this material.

Since the regulatory criteria will require an expanded sediment sampling, it will be necessary to standardize the methods of sediment collection, sample handling, and analysis to insure comparability of data for the regulatory decision-making process. To achieve this objective, the Great Lakes Laboratory has been working with the Environmental Protection Agency/Corps of Engineers Technical Committee on Criteria for Dredged and Fill Material to develop state-of-the-art guidance for the collection, handling, and analysis of sediment samples. This paper will discuss the present limitations on providing the required guidance.

PLANNING A SEDIMENT COLLECTION PROGRAM

An individual responsible for implementing a sediment sampling study is faced with numerous decisions. However, it is not possible to provide definitive guidance for all these decisions because of site-specific influences. This can best be illustrated by listing the decisions to be made and the factors that can influence each decision, as in Table 1. This approach clearly identifies those decisions related to sample collection as the most subjective. The next most subjective decisions are related to the selection of specific tests and analyses to be performed. While procedures and analytical techniques can be specified, the need and appropriateness is site- and project-dependent. The least subjective of the decisions relates to sample storage and handling as these procedures become mandatory once the tests and analyses are specified.

The summation in Table 1 also points out the interaction between the various decision points. For example, sampling locations, testing procedures, and specific analyses all depend on the specific purpose of the study. In addition, although testing procedures and analyses are chronologically last in the processing of samples, a decision on which test and analyses to perform must be made prior to sampling so that proper precautions can be taken for sample handling and storage.

It should be apparent that an essential component of any field sampling program is a pre-project meeting with all concerned personnel. The purpose of this meeting should be to define the objective of the sampling program so that the information needed can be matched with the specific tests that are available for use. This approach will have the effect of directing the sampling program at a specific need and assist in deciding which tests to perform. A second benefit of such a meeting is increased communication between participating groups so that field personnel are aware of potential sample handling and contamination problems and laboratory personnel are available to analyze collected samples within prescribed time limits.

TABLE 1. REQUIRED DECISIONS IN PLANNING A SEDIMENT SAMPLE COLLECTION PROGRAM

Decision Point	Influencing Factor
Where to sample	Major point sources Local hydrology Purpose of study
How many samples	Site variability Required degree of definition Resources (dollars and/or manpower) available
When to sample	Are seasonal fluctuations expected?
Type of sample	Purpose of study Tests to be run
How to sample	Site variability Purpose of study
How to store samples	Type of test to be run Type of analysis to be run
What tests to be run	Purpose of study
What analysis to perform	Site-specific factors Purpose of study

Any definition of a project should avoid generalized tasks such as "an environmental assessment of a proposed dredged material disposal operation". Although an environmental assessment may be the overall objective of the study, this objective should be considered a cumulative effect. Therefore, the objective of the sampling program should be subdivided into specific tasks such as:

a. Compare two or more sites in a project area.

b. Quantitate the total amount of certain contaminants present.

c. Determine the mobility of contaminants in dredged material.

d. Determine the distribution of certain chemical contaminants in the sediments of a project area.

e. Determine potential sediment toxicity.

f. Determine the biological suitability of project-site water.

g. Determine whether a local discharge has altered the water and/or sediments in the project area.

h. Determine the sediment-phase distribution of certain chemical contaminants in the sediments of a project area.

As will be stated shortly, the chemical tests available for sediment studies measure specific characteristics of sediments. Thus, the more explicitly the goals of a project can be stated, the easier it should be to select the tests to be run and, subsequently, the required method of sample handling.

Sample Testing Procedures

The next step in the project-planning process should be the selection of testing procedures to be used. Presently, the three types of tests available for the analysis of sediments include (1):

a. Bulk Analysis
b. the Elutriate Test
c. fractionation/extraction procedures

The utility of any one sediment test for a particular project can only be determined after the purpose of the study has been identified since each test provides different information as indicated below:

a. Bulk Analysis provides an estimate of the total concentration of a constituent in the sediment sample. The analytical result will include the various sediment phases (interstitial water phase, exchangeable phase, residual phase, etc.), but is poorly related to the biological availability of the constituent. A beneficial aspect of this test is that storage and preservation problems are minimized since changes in the oxidation state generally do not affect total concentrations. Bulk Analysis results are useful for calculating an inventory of the total amount of a constituent involved in a dredging project. However, a major limitation of the test is that results are a poor indicator of potential environmental effects of moving the sediments (2) (as in a dredging operation) because of the poor relationships between total concentration of a constituent in sediments and biological availability (3, 4) or water quality changes (5).

b. The Elutriate Test provides an estimate of the mobility of chemical constituents from the sediment phase to the water phase. This test has the advantage of being more environmentally interpretable since it measures "water soluble" constituents which are the basis of most water quality criteria. The disadvantages associated with the Elutriate Test are the lack of understanding of the mixing process that influences data interpretation (6), the fact that the test is of short duration and may not estimate long-term changes following disposal, and the fact that this test, like Bulk Analysis, does not address possible impacts on benthic fauna. In addition, the test requires a greater effort for storage and preservation of samples since oxidation state changes may alter test results (7).

c. The fractionation procedures provide more detailed information on the distribution of chemical constituents within the sediments by subjecting the sample to a series of increasingly harsh extractions solutions (5, 8). It is possible there may be a crude inverse relationship between the harshness of the extraction solution and the bioavailability of the constituents. However, the full meaning of a given distribution is not understood. Further limitations of this procedure are that the actual testing is more time consuming and strict storage requirements are mandatory.

An important aspect of the description of each test procedure is that each test measures a different property of the sample. The Bulk Analysis Test measures the total concentration of a chemical constituent of sediments regardless of chemical form and the Elemental Partitioning procedure measures the distribution of the total amount of material among several operationally-defined phases. The Elutriate Test measures chemical mobility under specified conditions. It is these properties, chemical presence, chemical distribution, and chemical mobility, that should be matched with the identified purpose(s) to determine whether the test is appropriate for the study in question.

A major limitation of all three tests is the present inability to interpret the results, particularly in terms of biological impact. For example, Bulk Analysis tells one how much is present, but the literature in the water field clearly demonstrates that total concentration is not the factor that determines biological response (2). Thus, any total measurement will overestimate potential biological response. Similarly, the Elutriate Test has some merit because the mobility phase is operationally equated with the soluble phase and the soluble phase has been classically used in the establishment of water quality criteria. However, water quality criteria have an implied time factor of 96 hours to one year whereas the Elutriate Test concentration represents end-of-the-pipe concentrations that are diluted by several orders of magnitude in 10 to 60 minutes. Because the exposure time and exposure concentrations are over exaggerated, the Elutriate Test is also a poor estimator of biological response (6). The Elemental Partitioning procedure is a relatively new test that has been used more as a research tool than a regulatory tool. Studies have indicated a time-dependent relationship between the more labile Elemental Partitioning phases and water quality changes (5), but no relationship between phase distribution and biological response (3). At present, the meaning of an elemental distribution is unknown.

A second limitation of all three tests is that each contains a degree of subjectivity. The Elutriate Test (1) consists of shaking one volume of sediment with four volumes of water for 30 minutes. Both the ratio and time are subjective. The Elemental Partitioning procedures described by Brannon et al. (5, 8) consist of the use of six extraction solutions. The number and relative strength of the extraction solutions are subjective. Finally, even Bulk Analysis procedures can be less than objective. Numerous digestion procedures have been described and the efficiency of any one digestion method can be affected by (1) length of digestion time, (2) the specific chemical contaminant of interest, (3) the type of sample, and (4) particle size (9, 10).

Any attempt to make the testing procedures more objective would be desirable from a scientific point of view, but would not necessarily be helpful from a regulatory point of view. The reason for this situation is that altering the testing procedure will not improve the ability to interpret the results. Thus, the present emphasis should be to apply the available tests, subjective though they may be, in a standardized manner.

Type of Samples

Once the tests to be used have been selected, the type of samples that have to be collected has been determined. When the Elutriate Test is to be run, both water and sediment must be collected. For the Bulk Analysis and Elemental Partitioning procedures, only sediment samples are needed.

Specific Analyses

The next item on the planning meeting agenda should be the selection of the specific chemical analysis to be completed. This is necessary to insure that proper handling and sample storage techniques are available during the collection period. It will also insure that sufficient sample is collected to perform all scheduled analyses.

The preparation of a mandatory analysis list should be discouraged. The analysis to be performed should be site specific and based on project purposes, major point sources, and/or local activities. For example, if the purpose is to clean up an industrial spill for PCBs, there is certainly reason to analyze for PCBs, but additional analysis for a wide spectrum of metals would not be useful in terms of the stated project purpose. This is not to say that sediment data is not scientifically useful, but rather a distinction between regulatory need and research must be made. Such a distinction will make most efficient use of the analytical dollars for specific projects.

Sample Storage Procedures

Specific identification of project purposes will lead to the selection of chemical tests to be used and chemical analyses to be performed. These decisions will simplify the selection of sample storage procedures. The use of the Elutriate Test and/or Elemental Partitioning will require samples to be stored moist in an oxygen-free environment. Bulk Analysis samples may be stored under a variety of conditions depending on the specific analyses to be performed. Different storage techniques are required because each test measures a different property of the sediment sample.

The Elutriate Test measures chemical mobility under specified conditions. Since mobility is a function of chemical speciation, any preservation technique must minimize changes in chemical speciation. Because oxidation will change chemical form, exposure to the atmosphere is undesirable. Chemical additives are also undesirable since they may alter solubility and chemical form (and, hence, mobility). The recommended method of sample storage is to approximate field conditions (refrigerate and seal in an oxygen-free environment) and process as soon as possible.

A similar approach of sample refrigeration in an oxygen-free environment is also recommended for Elemental Partitioning samples. Whereas the Elutriate Test is concerned about the "concentration" of a chemical constituent in a specific phase (water leachable), Elemental Partitioning is concerned about the distribution of a chemical constituent between several phases. Therefore, changes in oxidation and solubility are equally undesirable. Since there is no known preservative that preserves chemical distribution in sediments, the best state-of-the-art approach is to refrigerate the sample to inhibit bacterial growth and reduce Eh stress by isolating the sample from atmospheric contact. Freezing and drying are not recommended because physical changes can alter chemical form and, hence, distribution.

The third test procedure, Bulk Analysis, measures total concentration and is, therefore, less sensitive to chemical speciation changes. Consequently, greater flexibility in sample storage is acceptable for this procedure. The procedure used for the Elutriate Test and Elemental Partitioning can be used for Bulk Analysis samples, but freezing and drying can also be utilized for sample storage as long as the constituent of interest is not altered or lost by volatilization during the drying or thawing steps. A recommended list of analysis that can be performed on moist, dried, and frozen samples is presented in Table 2. It is apparent that the largest number of analysis can be run on moist samples since samples stored in this manner are least subject to change.

Information on sample handling for each chemical parameter was summarized in the format shown in Figure 1. The most extensive information was available for handling and storage of water samples. The least information is available for sediment sample handling. It was assumed that container requirements for water samples were also valid for sediment samples. A major unknown is the length of time that sediment samples can be stored prior to analysis.

SAMPLING CONSIDERATIONS

The remaining decision points in Table 1 relate to location, numbers, and method of sample collection. Each component is important because the quality of any sediment evaluation study is only as good as the information gained through sampling. Thus, any errors incurred during sampling will manifest themselves by limiting the accuracy and/or appropriateness of the study.

The objective of the sampling program should be to obtain representative samples using appropriate sampling techniques (11, 12, 13). However, despite the fundamental importance of these factors, it is difficult to provide specific guidance to achieve this objective because of site-specific factors such as major point sources and project purposes. Therefore, the best state-of-the-art guidance that can be provided is to be on guard against introducing a bias during sample collection (11).

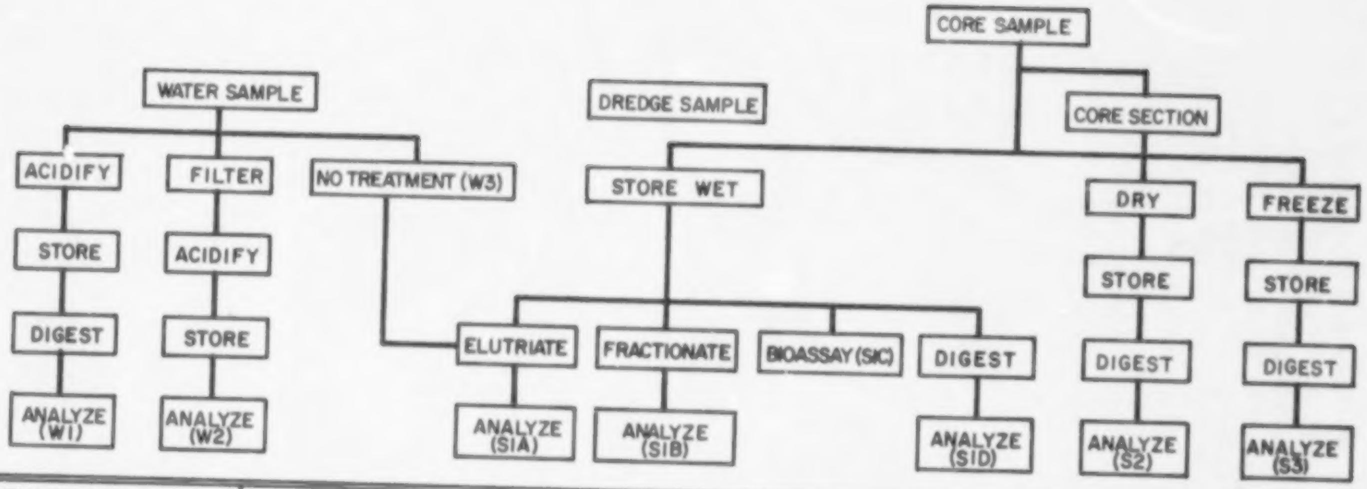
One potential bias can be introduced during the selection of sampling locations. For example, it is a usual practice to collect samples in the vicinity of major point sources and in quiescent areas that are conducive to the settling of finer-sized material. The former sites would be expected to have higher concentrations because of their proximity to the sources. The

TABLE 2. ACCEPTABLE METHOD OF SAMPLE STORAGE AS A FUNCTION OF BULK SEDIMENT ANALYSES TO BE PERFORMED

Wet		Dry	Freeze
CEC	ORG-P	Particle Size*	Particle Size*
Cl_2 Demand	Total-P	Minerology	Minerology
BOD	PAH	TOC	TOC
COD	Hg	TIC	TIC
SOD	Al	PCB	O & G
Carbamates	As	Pesticides	PCB
pH	Cd	ORG-P	Pesticides
SRP	Ca	Total-P	Phenolics
Redox	Cr	PAH	ORG-P
Total Solids	Cu	Hg**	Total-P
Volatile Solids	Fe	Al	PAH
Sulfides	Pb	As	Hg
Phenoxy Acids	Mg	Cd	Al
Particle Size	Mn	Ca	As
Minerology	Mo	Cr	Cd
TOC	Ni	Cu	Ca
TIC	Se	Fe	Cr
Pesticides	Zn	Pb	Cu
Phenolics		Mg	Fe
Spec. Grav.		Mn	Pb
NH_3		Mo	Mg
NO_2^-		Ni	Mn
NO_3^-		Se	Mo
ORG-N		Zn	Ni
TKN			Se
O & G			Zn
PCB			

* Dispersed particle size probably not affected by drying or freezing. Apparent particle size may be affected.

** Mercury may be lost if sample is dried at too high a temperature.



SAMPLE DESIGNATION	W1	W2	W3	SIA	SIB	SIC	SID	S2	S3
PURPOSE	TOTAL WATER CONCENTRATION	SOLUBLE WATER CONC.	USED IN ELUTRIATE	MOBILE CONC.	CHEMICAL DISTRIBUTION	BIOAVAIL-ABILITY	TOTAL SEDIMENT CONC.	TOTAL SEDIMENT CONC.	TOTAL SEDIMENT CONC.
CONTAINER	G,P	G,P	G,P	G,P	G,P	G,P	G,P	G,P	G,P
SAMPLE TREATMENT	NONE	FILTER	NONE	NONE	NONE	NONE	NONE	AIR DRY	FREEZE
PRESERVATIVE	HNO_3 pH<2	HNO_3 pH<2	NONE	4° C (MINIMIZE	4°C AIR CONTACT	4°C KEEP FIELD MOIST.)	4° C	NONE	NONE
STORAGE TIME	90d	90d	<1w	<1w	<1w	<1w	<1w	–	–
DIGESTION SOLUTION	STRONG ACID	NONE	–	W3	VARIABLE	–	STRONG ACID	STRONG ACID	STRONG ACID
SAMPLE VOLUME OR WEIGHT	100-500 ml	100-500 ml	VARIABLE	VARIABLE	300-500g	VARIABLE	2-5g	2-5g	2-5 g

Figure 1. Schematic diagram for processing samples scheduled for metal analysis.

latter sites would also be expected to have a high concentration because the smaller particles that accumulate in quiescent areas are known to concentrate many chemical contaminants (14, 15). Thus, this approach would be expected to produce a result that is biased high. While this result would be acceptable if the purpose is to locate the maximum concentration in a project area, it would produce non-representative results if the purpose is to define the distribution of chemicals within a project area. Suggestive guidance in the latter case would be to divide the sampling locations between the anticipated maximum concentration zones and the remainder of the project area. The important point is that both the project site and project purposes should be included in the selection of sampling locations.

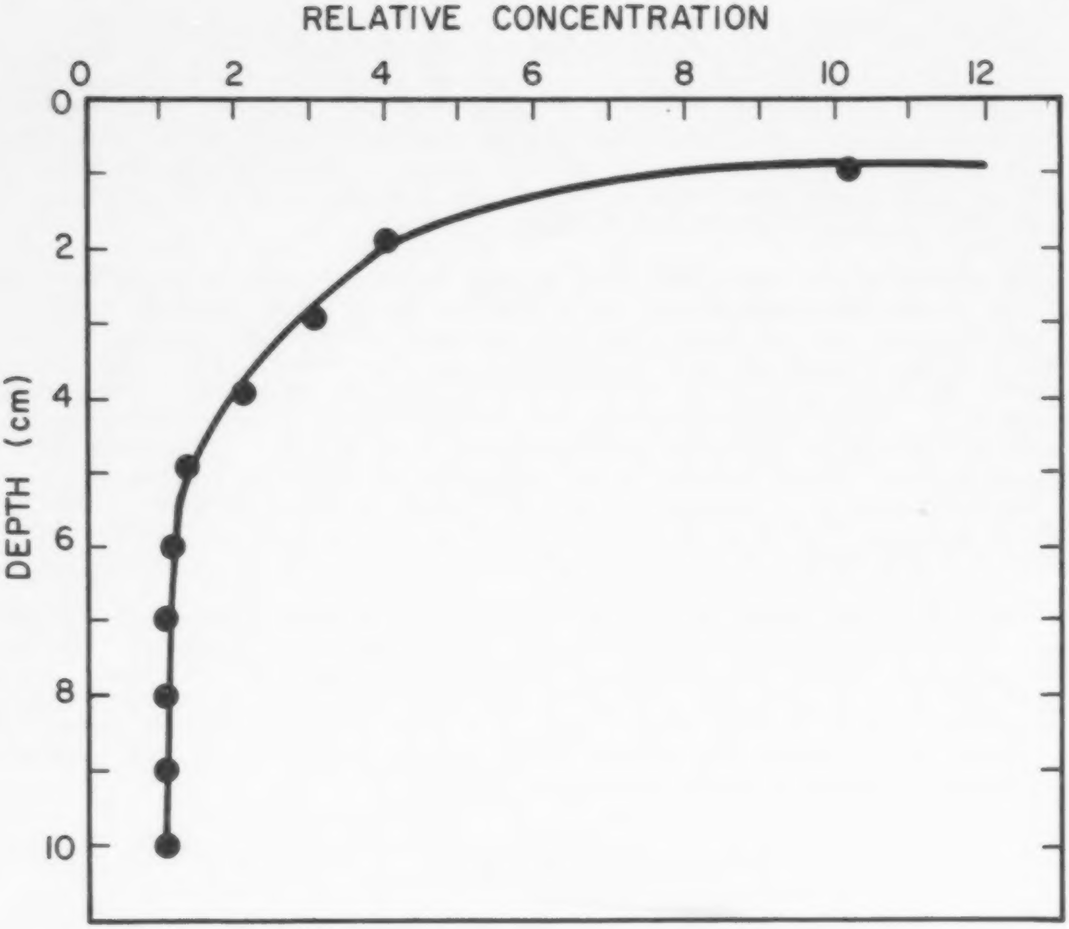
A project manager should also be aware of the fact that the selection of sampling equipment can have a potential effect on resultant data. The choice is restricted to dredges and corers and dredges are frequently used because of ease of operation and larger sample recovery. However, since the depth of sediment penetration by a dredge can be affected (16, 17) by the weight and shape of a sampler, the height of free-fall, the angle of impact, and sediment texture and density, the recovered sample is site-dependent. Because sediments are frequently stratified in the vertical dimension and the depth of sediments sampled is variable, grab samplers may introduce analytical variability into the final data that is a function of dredge penetration rather than being a property of the sample. This effect is summarized in Figure 2 for a hypothetical situation where the sediment concentration varies by a factor of 10. In an extreme case, a differential dredge penetration could produce more than a 300 percent variation in analytical results at the same sampling location.

The need to consider this artifact of sampling may vary with project purposes. However, when important, corers should be the method of choice in areas that sediments are known to be stratified. Dredges should be used in areas where sediments are known to be homogeneous and differential penetration would not have an effect on sample composition.

The final point to be mentioned relates to the number of samples to be collected. Again, it is not possible to provide specific guidance on this point because of site-specific variability and project-specific needs. However, it should be pointed out that the number of samples will be proportional to the financial resources available and inversely proportional to the analytical cost per sample:

$$\text{Number of samples} - \frac{\text{Dollars for analysis}}{\text{Cost per sample}}$$

This relationship suggests that a judicious selection of parameters to be analyzed is required. If a large number of analyses are to be completed on each sample, the cost per sample increases and the number of samples, therefore, has to decrease. An awareness of this fact reinforces the position stated earlier that a mandatory list of analyses should not be prepared.



DEPTH OF SAMPLE COLLECTED	CALCULATED AVERAGE CONCENTRATION
1	10
2	7
3	5.6
4	4.7
5	4.0
6	3.5
7	3.1
8	2.9
9	2.7
10	2.5

Figure 2. Effect of vertical sediment profiles on grab sample composition as a function of penetration.

SUMMARY

Recent regulatory developments will require a more systematic testing of sediments to be dredged. Based on this need, the Great Lakes Laboratory is working with the U.S. Environmental Protection Agency and the U.S. Army Corps of Engineers to develop a sediment collection and analysis manual. This experience indicates that it is possible to provide guidance on how to perform available testing procedures and specific analyses. However, the decisions on where to sample and when to use the testing procedures is site-specific and the responsibility of the project manager.

A procedure is described that allows a project manager to assess the utility of available procedures for a specific study. This approach relies on an awareness of the sediment properties being characterized and the relation between these properties and the purpose of the study. The selection of specific chemical analysis to be performed should also be based on the purpose of the study. Once the procedures and specific analysis have been selected, the method of sample handling and storage will be defined. The least objective portion of the procedure is the selection of sampling locations because of the importance of site-specific variability and project purposes. The best general guidance that can be provided is to guard against the introduction of a bias as a consequence of selecting the sampling locations.

Each test procedure considered measures a different property of the sample. The tests are complementary in that they measure the presence, distribution, and mobility of sediment-associated contaminants. Unfortunately, these properties are not directly related to the potential environmental impact of sediment-associated contaminants. Thus, the major state-of-the-art limitation in conducting a sediment evaluation is the absence of valid criteria to evaluate results in response to the regulatory mandate.

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